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Abstract

Agricultural expansion and intensification drive the conversion of natural areas worldwide. This trend will likely continue, particularly in South America, as rising global population, dietary shifts and the increasing importance of biofuels will further accelerate the demand in agricultural products. Yet, it is not clear where and how much production would need to expand and intensify to meet future demands and how policies may help minimizing environmental trade-offs. Particularly the latter requires an understanding of the underlying forces that drive agricultural land-use changes and how they play out given different spatial chracteristics of regions. In concert with scenario analyses, this offers a framework for planners and decision makers to explore potential impacts from policies, especially in very dynamic regions. Argentina, where agricultural expansion and intensification result in dramatic conversions of natural areas, is a good example of a dynamic human-environment system. The overarching goal of this thesis was to understand the drivers of agricultural land-use change and to explore future trajectories of land-use change, and how economic and conservation policies may impact them in Argentina's most important agricultural areas. First, this thesis examines drivers of agricultural land-use changes using a net returns model of agricultural production. Then, this thesis evaluates the effects of economic and conservation policies on future land-use changes and on the connectivity of forests by developing scenarios of future land-use change. The results highlight that agricultural intensification in Argentina is driven by economic interventions, whereas agricultural expansion primarily responds to environmental characteristics and zonation programs. In addition, economic policies may have less power in governing land use changes than previously thought, as results suggest that there are other factors, than profit maximization, influencing land conversions. Future agricultural development would occur in priority areas for conservation in Argentina, but zonation policies, such as the Forest Law, appear to be powerful in limiting potential environmental trade-offs. Results also show that conservation planning does not necessarily need to conflict with economic development in Argentina, since under similar deforestation rates; landscape planning can preserve forest connectivity in the Chaco. Overall, this thesis highlights that a-priory evaluation of potential future effects of economic and conservation policies on land-use change can help informing spatial and conservation planning to steer development pathways towards desired directions in dynamic agricultural regions.

Zusammenfassung

Landwirtschaftliche Expansion und Intensivierung treiben die Umwandlung natürlicher Ökosysteme weltweit. Dieser Trend wird sich voraussichtlich fortsetzen, vor allem in Südamerika. Die wachsende Weltbevölkerung, Ernährungsumstellungen und die zunehmende Bedeutung von Biokraftstoffen wird die Nachfrage nach landwirtschaftlichen Produkten weiter steigern. Das Ausmaß und die räumliche Verteilung landwirtschaftlicher Expansion und Intensivierung sind bis heute nicht absehbar. Darüber hinaus ist es unklar, inwieweit politische Maßnahmen negative Folgen für die Umwelt minimieren können. Gerade Letzteres erfordert eine Untersuchung der zugrundeliegenden Prozesse, die die landwirtschaftlichen Expansion und Intensivierung antreiben und wie sich diese unter heterogenen naturräumlichen Eigenschaften verhalten. In Kombination mit Szenarien-Analysen kann ein Rahmen zur Unterstützung von Planungsprozessen geschaffen werden, um potentielle Auswirkungen von politischen Maßnahmen – insbesondere in sehr dynamischen Regionen - zu erforschen. Argentinien ist geprägt von dramatischen Umwandlungsprozessen natürlicher Ökosysteme durch landwirtschaftliche Expansion und Intensivierung und ist damit beispielhaft für das dynamische Zusammenspiel von Mensch und Umwelt. Das Hauptziel dieser Dissertation war es, die Triebkräfte der Veränderung von Argentiniens Agrarlandschaften zu verstehen, potenzielle zukünftige Landnutzungsveränderungen zu analysieren und den Einfluss ökonomischer und naturschutzbezogener politischer Maßnahmen auf diese zu erfassen. Im ersten Teil der Dissertation wurden die Triebkräfte landwirtschaftlichen Landnutzungswandels mittels eines Nettoertrags-Modells ermittelt. Anschließend wurde der Einfluss von ökonomischen und naturschutzbezogenen Maßnahmen auf zukünftige Landnutzungsveränderungen, sowie auf die Konnektivität von Waldgebieten mit Hilfe von Landnutzungs-Szenarien analysiert. Die Ergebnisse dieser Dissertation zeigen, dass landwirtschaftliche Intensivierung von ökonomischen Maßnahmen getrieben ist, während landwirtschaftliche Expansion hauptsächlich durch naturräumliche Eigenschaften und Zonierungsprogramme

determiniert wird. Diese Arbeit zeigt weiterhin, dass Faktoren jenseits der Profitmaximierung solche Umwandlungsprozesse treiben. Daher haben ökonomische politische Maßnahmen möglicherweise einen geringeren Einfluss auf Landnutzungswandel als bisher erwartet. Die zukünftige Entwicklung von Agrarland konzentriert sich räumlich auf Gebiete mit hoher Priorität für den Umweltschutz. Zonierungsprogramme wie das Argentinische Waldgesetz stellen demgegenüber wirkungsvolle Maßnahmen dar, um umweltschädigenden Entwicklungen vorzubeugen. Die Erkenntnisse aus Argentinien zeigen, dass Naturschutz nicht zwingend im Konflikt mit ökonomischer Entwicklung steht, denn mittels Landschaftsplanung kann die Konnektivität von Waldgebieten auch unter gleich bleibenden Abholzungsraten bewahrt werden. Zusammenfassend zeigt diese Dissertation den großen Mehrwert von a-priori Evaluierungen der potentiellen Einflüsse ökonomischer und naturschutzbezogener Maßnahmen auf Landnutzungswandel. Die daraus gewonnenen Informationen können räumliche Planungsprozesse unterstützen und helfen die Entwicklung dynamischer Agrarlandschaften besser zu steuern.

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Chapter I:
Introduction

1 Scientific background

1.1 Global land-use change and human-environmental systems

For more than 10,000 years, land use has played a crucial role in the development of human societies. Humans rely on agriculture and forestry for obtaining food, fibre, and bioenergy in order to satisfy livelihood demands (MEA, 2005). However, these human activities have already modified 75% of the Earth's ice-free terrestrial surface as a result of residence or land uses (Ellis and Ramankutty, 2008). Of all the land under human use, agricultural activities occupy more than a third of the Earth's surface (FAOSTAT, 2010) (12% cropland and 22% pasture lands (Ramankutty et al., 2008)) while less than a quarter remains as natural areas (Ellis and Ramankutty, 2008).

With technological developments, the expansion and intensification of agriculture has increased agricultural production, thus feeding a growing population that increasingly demands richer, more resource intense diets and that will continue do so in the future (Foley et al., 2011; Smith et al., 2010; Tilman et al., 2011). An increased share of agriculture in a region provides employment, increases national capital and boosts the service sector (Thornton, 1973). An increment in small-holder agriculture can reduce hunger and alleviate poverty in rural local communities, to the point that developing nations have focused their development plans on increasing primary productivity to improve rural development (Anríquez and Stamoulis, 2007). Traditional agricultural systems (i.e., agroecosystems) also hold important historic crop varieties and conventional ways of working the land that have positive repercussions on the preservation of ecosystem services (such as the regulation of soil and water quality, or carbon sequestration) and species that benefit from heterogenous agricultural systems (Power, 2010).

The intensification of agriculture is associated with the usage of high amounts of inputs, such as fertilizers or herbicides, which can impact the health of local livelihoods and pollute the environment (Kirkhorn and Schenker, 2001; Tilman, 1999). On the other hand, the expansion of agriculture at a large scale translates into degrading and declining of provisioning and non-provisioning ecosystem services, thus challenging the resilience of animal species (Hansen et al., 2013; Maxwell et al., 2016; Sanderson et al., 2002) (Figure I-1). More specifically, agricultural expansion has been responsible for the clearance of about three quarters of the world's forest (Kissinger et al., 2012), is the second largest global threat for biodiversity conservation (Maxwell et al., 2016) and one of the main causes of ecosystem fragmentation (CBD, 2010; Sala et al., 2000) among other environmental impacts (Baudron and Giller, 2014) (Figure I-1).

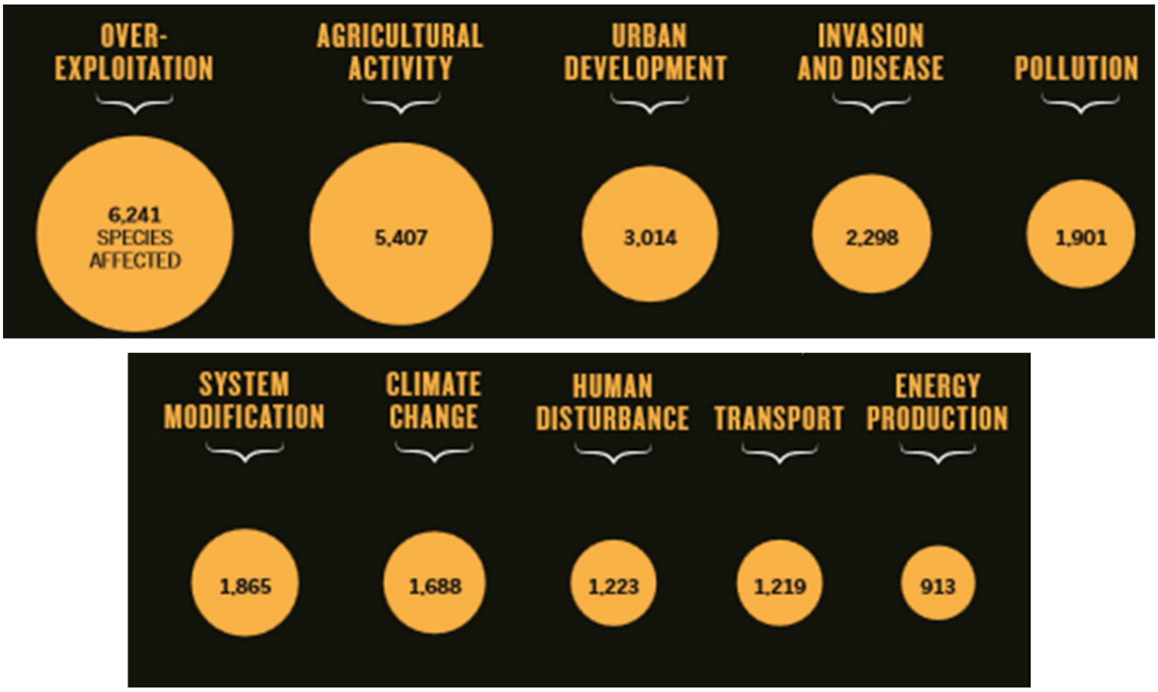

Figure I-1: Number of IUCN (International Union for Conservation of Nature) RED list species threatened by human activities (adapted from Maxwell et al. 2016).

In recent decades, the concentration of resources (i.e., land) and technology have landed in fewer hands (i.e., producers tend to be big international companies) (Borras Jr and Franco, 2012). This can aggravate inequality between prosperous and not so prosperous nations (Muradian and Martinez-Alier, 2001) and can result in often highly-productive agricultural regions exporting their produce, while at the same time importing the socio-environmental externalities of intensive production resulting in social and environmental conflicts (Smith et al., 2010). Moreover, large-scale industrial crops are often redirected towards feeding the livestock sector or meeting biofuel mandates in more prosperous countries – exacerbating issues of food access and food security.

From a global and market-based perspective, agricultural activities are often regarded as a form of investment and sometimes lead to the speculation of land for commodities production (Chodorov, 1960). This means that future agricultural developments are in the hands of a few entrepreneurs (i.e. large scale land acquisitions) with thin ties to the land that often impose productive structures that may be in conflict with local dynamics, often generating additional social and ecological impacts (Borras Jr and Franco, 2012). This is of special importance in developing regions and poses several challenges to the future of agriculture where rich biodiversity coexists with weak planning regulations and enforcement (Nolte et al., 2017b). Understanding how and why agricultural conversions occur, and how they impact the environment is thus essential to avoid unwanted outcomes from spatial and conservation planning.

Specifically, to help informing land-use and conservation planning to avoid unwanted environmental impacts, three main knowledge gaps must be addressed: First, gain a better understanding of underlying forces of land conversions that drive the demand for agricultural land (i.e., underlying drivers) and the spatial characteristics that determine the allocation of the resulting agricultural land-use (i.e. spatial determinants). This is crucial in order to identify policy levers that can influence land-use change towards desired outcomes (Angelsen, 2010; Meyfroidt et al., 2014). Second, understanding land-use pathways and potential future agricultural trajectories can elucidate on potential future land-use conversions (by means of scenario analysis) to evaluate the impact that specific policies can have on the environment (Martinuzzi et al., 2015; Neumann et al., 2011). Decision makers can then better plan for future spatial developments. Third, assessing potential environmental impacts from future agricultural trajectories can exemplify ex-ante possible policy outcomes and thus allow to re-design policies and take decisions in an informed way (Helming, 2014; UNDP, 2009).

1.2 Main pathways and impacts of agricultural land-use dynamics

There are two main pathways to increase agricultural output in order to meet rising future demand. First, by expanding agricultural activities into natural areas, as in case of the tropics where between 1980 and 2000, 80% of agricultural expansion has been at the expense of primary or secondary forests (Gibbs et al., 2010a). Agriculture expansion fragments, degrades and depletes ecosystems (CBD, 2010; Fahrig, 2003; Sala et al., 2000) with severe negative effects on mammals, birds, reptiles, amphibians and insects (Koh and Wilcove, 2008; Ripple et al., 2014; Whittaker et al., 2013). Species experience even stronger negative effects in fragmented landscapes under climate change conditions, because adaptation (e.g., via range shifts) may be impossible (Brook et al., 2008; Travis, 2003) since changing climatic conditions can restrict and disconnect the habitat of specialist species , impeding their migration and isolating populations to their decline (Wade et al., 2003). Nowadays, only 4,000Mha of non-cropland worldwide can be used for expanding rain fed agriculture (Lambin and Meyfroidt, 2011) and thus the location of agricultural expansion to meet potential future production demands is restricted and can have devastating impacts if concentrated in sensitive environmental regions.

Second, increases in agricultural production can be reached by intensifying agricultural activities by changing agricultural inputs or practices (Erb et al., 2013; Kuemmerle et al., 2013). For example, increasing the use of fertilizer, pesticides or machinery, modifying production systems from traditional ranching or cropping characterized by high labor input and little mechanization towards capital-intensive large scale commercial cropping, and switching animal production from traditional staple systems towards highly intensive feedlot systems. Although we would expect that increasing production output per unit of land would translate into less land brought into production, increasing global demand for agricultural products has led to increasing marginal commodity prices which motivates new investments into agricultural activities and thus agricultural expansion into natural areas (Angelsen and Kaimowitz, 2001). This rebound effect of land-use expansion under an intensified used can be exemplified by the Jevon’s paradox (Jevons, 1866), in this case, when the marginal revenues of an activity are then used to deforest and expand agricultural activities somewhere else, potentially jeopardizing the conservation of natural systems (Angelsen and Kaimowitz, 2001; Macedo et al., 2012; Phelps et al., 2013). The environmental impacts of agricultural intensification on natural systems include the eutrophication of waters due to increase fertilizer application or soil depletion due to the avoidance of fallow cycles to increase production (Cunningham et al., 2013; Foucher et al., 2014). These environmental impacts have a negative effect on biodiversity: birds, mammals and amphibians are often threatened by the effects of agricultural intensification (Gibbs et al., 2009; Hof et al., 2011; Kleijn et al., 2009) to the point that the diversity of species can be affected (Newbold et al., 2015).

However, there are integrative solutions to balance agricultural production systems with the preservation of biodiversity and ecosystems functions. For example limiting crop waste by using pest control mechanisms, improving transportation systems (Bruinsma, 2009), using genetically modified organisms, and generally improving the efficiency of agricultural practices. Through such practices the negative impacts of land-use intensification and agricultural land expansion could be minimized while at the same time closing yield gaps (Carberry et al., 2013; Laurance et al., 2014; Mueller et al., 2012). Other improvements may be possible.

Nonetheless, not all regions in the world can easily increase yields in a sustainable way and, in fact, some world agricultural areas are reaching their yield production limits (Ray et al., 2013). This means that if regions fail to increase yields, the expected increase in agriculture produce demand may be satisfied by means of agricultural expansion (Tilman et al., 2011) and caution is needed not to incur in negative impacts on biodiversity and ecosystems. Besides, the impacts of the expansion and intensification of agriculture can be combined and become even more dramatic than those from isolated conversions. Therefore, balancing agriculture production with nature preservation meanwhile minimizing trade-offs possess challenges for future land development and management strategies (Chaplin-Kramer et al., 2015; Grau et al., 2013).

1.3 Drivers of agricultural land-use change: underlying causes and spatial determinants

Individual farmers, agricultural enterprises, and governments are the decision makers of land conversions. Whether they convert natural land to agricultural use or intensify existing agricultural land depends on a range of factors operating at different levels from the global to the local level (Angus et al., 2009; Reidsma et al., 2006). The underlying causes that drive land use change, are factors or reasons that affect an individuals' decisions on changing land uses, and can be broadly grouped into demographic (e.g., population growth, aging population), economic (e.g., agricultural commodity prices), technological (e.g., innovations), institutional (e.g., taxation, land-use or conservation policies), and cultural factors (e.g., religion) (Geist et al., 2006; Lambin et al., 2003). While in the past, many of the underlying causes of agricultural land-use change have operated at regional-to-national scales; increasingly many of these factors are becoming global in scope. For example, global prices for agricultural commodities can impact land-use change across the globe, and at times dwarf the impact of more local drivers of agricultural land-use change (Golub and Hertel, 2008; Liu et al., 2013).

This is further complicated by underlying causes typically playing out differently against the background of different environmental, socioeconomic, or historical spatial settings. Spatial determinants such as soil fertility, rainfall patterns, access to markets, or labor availability also impact land owner decisions and are important in allocating land-use change and thus explaining the spatial patterns of agricultural land-use change (Lubowski et al., 2008; Murgida et al., 2014). For example, in Argentina, a world producer of beef, cattle production is strongly influenced by a combination of local climate and global markets (Murray et al., 2016). Therefore, understanding how different regional and global factors interact with local site conditions to create the heterogeneous patterns of land-system dynamics around the world is important to identify effective policies that can adapt to changing future conditions (Angelsen, 2010; Geist et al., 2006; Levers et al., 2014).

1.4 Future land-use and conversion pathways

Future global demand and consumption of agricultural commodities and urbanization will influence the spatial patterns of land-use systems and the structure of the remaining cover and functioning of ecosystems. By 2050, global demand for crop is expected to grow by 100% (Tilman et al., 2011). It is important to understand the factors that influence future global productive land-use demands in order to inform spatial decision making and steer future policies to avoid unwanted outcomes from future land-use conversion. First, future population projections estimate 9.1 billion by 2050 (FAO, 2009) that would be concentrated in some of the countries with highest demand for imported agricultural produce (such as China or India). Second, changing nutrition trends towards more meat and dairy rich diets increase the pressure on agricultural production and thus on natural resources (Alexander et al., 2015; Alexandratos and Bruinsma 2012; Smith et al., 2010). Moreover, the increased usage of crops being used for animal feed or biofuel generation instead of directly for food (i.e., food vs. feed), increases local food prices, decreases local purchase power and hinders local food accessibility in some regions (Borras Jr and Franco, 2012). Third, the unbalanced distribution of food in the world and the lack of food sovereignty in some important agricultural regions, makes it more difficult for food to reach all people, and contributes to increasing overall food waste (in regions where food is abundant) and social conflicts, in areas of unbalance distribution of food (Tscharntke et al., 2012). The influence of these factors on future land-use patterns is unknown and possess important challenges since it is unknown how and where future agricultural developments, and its associated impacts on human-natural systems, will take place. To overcome this challenge, it is important to evaluate the range of potential effects of specific policies on land-use patterns to inform decision-making and overall, conservation and planning policies. Spatial tools combined with scenario analysis can be used for this purpose. For example, the usage of regression models can help us to disentangle the effect of underlying drivers of land-use change and when combined with scenarios, they offer the possibility to assess the potential future impact of each policy on landscape patterns.

Regression techniques are powerful tools to identify the factors that drive land-use processes. Econometric models of land-use change are one type of regression techniques that provide a framework for analyzing the relationships between drivers of land-use change and their impact on land conversions (Bockstael et al., 1995; Butsic et al., 2010; Lewis et al., 2008; Müller and Sikor, 2006). Based on theoretical models of human behavior, econometric models explain observed land-use change in terms of economic (including non-market) rents (i.e., profits). The basic intuition of these models is that individual land owners maximize the utility from land use (Capozza and Helsley, 1989). The output from these models are spatial statistical estimates of the effect of political, biophysical, and economic variables on the likelihood that a specific land-use change will occur. This theoretical model can then be translated to a statistical model via regression techniques (Wooldridge, 2011) to model observed land-use change in terms of economic (including non-market) returns (Irwin and Bockstael, 2004) and that is why they are also termed net returns models. Because econometric land-use models measure responses to economic returns, they provide a powerful tool for analyzing the effects political change and economic incentives have on land rents (Lubowski et al., 2006). Therefore, they can be used to understand the effects specific spatial and non-spatial policies can have on shaping future landscapes.

1.5 The role of scenarios in informing spatial planning and policies

Policies with a strong spatial component can influence the structure and patterns of landscapes and therefore the impacts land-uses can have on the environment (Brenner, 2011; Robinson et al., 2002). Therefore, informing decision makers about the potential effects that specific policies can have on the landscape and natural environments can be beneficial to steer development pathways as desired. There is a range of analysis that can provide objective information about the most important forces that drive land conversion and the associated impacts of policies on the environment. Among these analyses, the regression techniques described above can first be used to understand drivers of future developments and associated natural cover decline (Liu et al., 2016; Padeiro, 2016). Second, in a more participatory fashion, stakeholders' views can be integrated into idea-diagrams that can later be used to develop future scenarios in qualitative (i.e., participative qualitative scenarios) (Boron et al., 2016; Yang et al., 2016) or in quantitative ways (Martinuzzi et al., 2015; Tejada et al., 2016). The integration of statistical modeling with quantitative narratives of plausible futures (i.e., scenarios) that are translated into spatial maps of future land use, offer the most comprehensive set of characteristics with intuitive and practical interpretation of the impact of policies (Mallampalli et al., 2016).

Land-use scenarios are strategic tools that can be used to inform spatial planning in order to understand and visualize potential outcomes from specific spatial policies such as the imposition of a new tax, the construction of new roads, or the creation of new protected areas. Scenarios provide a background of alternative futures against which political strategies can be formulated and tested (Milburn, 2005). They do not aim to predict the future in terms of final land-use dynamics but they can serve to evaluate how much the system can potentially deviate from desired situations and the associated undesirable outcomes (Peterson et al., 2003) , which is highly useful for understanding coupled systems (Carpenter, 2002). The development of scenarios should be based on contrasting futures that reflect potential trajectories of land systems which can also include very unlikely situations that push the boundaries of common future assumptions (Gavier-Pizarro et al., 2014). Depending on the role they play in spatial planning, spatial scenarios can anticipate or explore potential future land-use conversions. Anticipatory scenarios, target development goals and implement plausible images of how the future will look like. Exploratory scenarios, perceive the final state of a land system as the result of the evolution of a plan (Mahmoud et al., 2009; Xiang and Clarke, 2003).

Regions with highest uncertainties in future development can benefit the most from the usage of scenarios because they offer solutions to complex issues for which there appears to be no simple analysis (Rounsevell et al., 2005). Among such regions, South America has been highlighted as a continent that will experience continuous future land-use conversions since there are still large reserves of cultivable croplands (Bruinsma, 2009; Ramankutty et al., 2002; Schmitz et al., 2014). Specifically, the tropical and semi-arid regions of South America may accommodate much of the foreseen land conversions in the future (Gibbs et al., 2010a; Laurance et al., 2014; Tilman et al., 2001). Yet, there are important knowledge gaps regarding the direction, magnitude and location of these potential future land-use changes and how policies can influence them. Therefore, understanding in which direction spatial policies can steer agricultural productive systems and their associated ecosystems is a current challenge that managers and policy makers face. In this regard, the usage of scenarios can aid in informed decision making to avoid unwanted and unforeseen outcomes from policies (Gavier-Pizarro et al., 2014).

1.6 Dynamics of contemporary commodity agricultural frontiers in South America

Agricultural expansion frontiers tends to occur where there is land available that offers profit to be earned (Barbier, 2012). According to the classic theory of von Thünen, land prices may decrease with increasing distance to infrastructure or towns, and with it, the willingness to invest in expanding agriculture due to a lack of capital availability (O'Kelly and Bryan, 1996). However, what currently can be observed is that agriculture expands in a way that does not respond only to von Thünen's theory, but also in regions far away from markets, infrastructure or cities. This is particularly true for regions in South America where remote tropical dry forests are being deforested at the highest world rates among tropical deforestation, mainly for the production of agricultural commodities (cash crops and livestock meat) (Gasparri and le Polain de Waroux, 2015; Hansen et al., 2013). Although these remote areas were originally settled by small farmers or not inhabited at all, government policies promoted the setting of large land holders by implementing strategic incentives (i.e., giving land rights or constructing infraestruture) (Barbier, 2014). In such contemporary commodity agricultural frontiers, new risk-prone investors that bid for the highest land rents, highly influence the expansion (by property aggregation) or sometimes the surge of new frontiers in more isolated regions (by using the profits from previous investments) (Le Polain de Waroux et al., in revison). Therefore, current agricultural dynamics have other factors influencing land-use conversions other than structural dependency and traditional economy theory and can bring uncertainties related to future agricultural dynamics.

An example of such contemporary commodity frontier in South America is Brazil. Between 1970 and 2011, Brazil lost 20% of its tropical forests and 40% of its dry savannas due to agricultural expansion (mainly for the production of cattle feed such as soybean and maize, and cattle ranching). This deforestation caused substantial releases of carbon from deforestation and agricultural practices (Galford et al., 2013) and significant biodiversity losses (Vieira et al., 2008). A second example is the Gran Chaco: a dry tropical forest, that lost 20% of its cover between 1985 and 2013 due to cropland and cattle ranching expansion (Baumann et al., 2016). This has translated into increasing carbon releases, fragmentation of dry forest ecosystems, and biodiversity changes (Baumann et al., 2016; Gasparri and Grau, 2009; Torrella et al., 2013; Torres et al., 2014).

Although there are examples of deforestation slowing down due to policies and social pressure, such as the case of the decrease in deforestation in the Brazilian Amazon (Nepstad et al., 2014) or the Atlantic forest of Paraguay (Baumann et al., 2017), this has translated into rapid deforestation in the neighboring Cerrado and Chaco ecoregions

(Baumann et al., 2016; Spera et al., 2016). Understanding therefore, what drives land-use changes in agricultural frontiers and to what extent they influence potential impacts on the environment is essential for making informed management decisions to avoid unwanted outcomes and leakage effects from spatial policies.

2 Conceptual framework

2.1 Study area and motivation

Main agricultural regions in Northern Argentina and their natural environment

Due to the Northern Argentina's unprecedented rates of deforestation, agricultural expansion and intensification, it is a prime study area with regard to understanding the drivers of land-use change, and designing effective and responsible land-use planning. Northern Argentina's grasslands and tropical dry forests are located in the Chaco, the Espinal and the Pampas ecoregions and covers an area of approximately 1.3 million km^2.

The Gran Chaco is a neotropical dry forest ecoregion that expands between Argentina (60%), Paraguay (28%), Bolivia (11%) and Brazil (1%) covering an area of more than 1 million km^2 (Figure I-2). The climate is semi-arid and highly seasonal, with a distinct dry season in autumn and winter (May–September), and a warm, wet season in spring and summer (November–April). The mean annual temperature is ~22°C, with an average monthly maximum of 28°C (Minetti, 1999). Annual precipitation ranges from 1,200mm in the east (wet Chaco) to 450mm in the west (dry Chaco). Elevation varies marginally except for the west and southwest of the study area where more hilly terrain prevails. Soils in the Chaco vary from being rich in minerals and fine in texture in the north (well-suited for agriculture) to the southwest of the ecoregion where soils are sandy with low content in organic matter - as in the center of the Espinal (Burkart et al., 1999). Natural vegetation in the Chaco consists of closed forest, open woodlands, shrublands, and palm savannas. Forests are the most characteristic vegetation formation and are typically dominated by species of the genera Schinopsis and Aspidosperma ("quebrachos") (Prado, 1993). This variety of environments enhances the rich biodiversity of this ecoregion which includes 145 mammal species (12 endemic), 409 birds (7), 54 reptiles (17), 34 amphibians (8), and more than 80 plant genera (3,400 species, of which 400 are endemic)(Bucher and Huszar, 1999; Giménez et al., 2011) . The Chaco is also a globally significant carbon pool (Baumann et al., 2016; Gasparri et al., 2008). Despite being a priority for biodiversity and ecosystem functioning preservation the Gran Chaco is in dire need of conservation action (Kuemmerle et al., 2017). Since 5000 years ago this ecoregion has hosted a great diversity of indigenous communities from the Andes, Pampas and Amazonia that were nomadic hunters and gathered forest resources (Brown et al., 2010). After the

Spanish invasions the usage of the land became more sedentary and intensive (with traditional cattle ranching and the introduction of cotton plantations) until today with unprecedented rates of land-use change for intensive cattle ranching and the production of large-scale cash-crop commodities (Baumann et al., 2016).

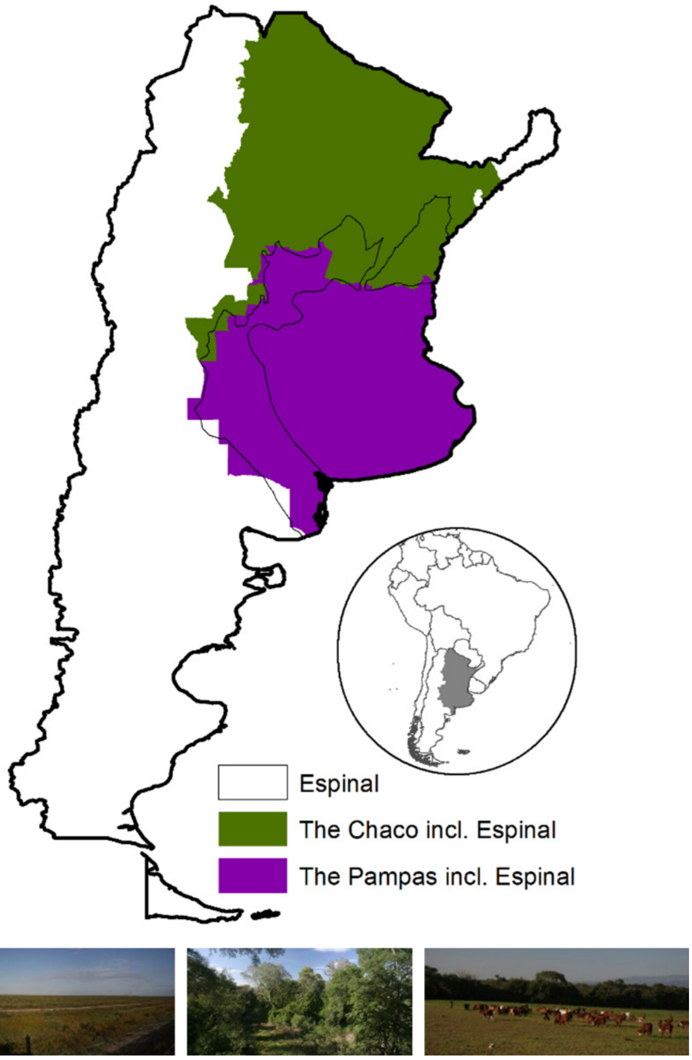

Figure I-2: Location of the Pampas, Espinal and Chaco ecoregions of Argentina within the departments of the study area of this thesis. Photos (left to right): soy field (T. Kuemmerle), Chaco forest (G. Gavier-Pizarro), ranching (M. Piquer-Rodríguez) 2012.

The Argentine Pampas are part of the Southern Cone grasslands of Uruguay, Paraguay, Brasil and Argentina (Herrera et al., 2014) (Figure I-2). This flat-terrain region has subtropical to temperate climate (mean annual temperature is ~15°C, with an average monthly maximum of 22°C and annual precipitation ranges 1,100mm in the east to 800mm in the west) (Bianchi and Cravero, 2010) and soils are very rich in organic matter (Herrera et al., 2014). The Pampas region, which is dominated by grasslands, mainly composed of *Stipa sp.*, *Briza sp.*, *Bromus sp.*, and *Poa sp.* (Cabrera, 1971), has experienced agriculture expansion since the early 19th century with a noted increase in production since the 1980's (Manuel-Navarrete et al., 2009). By the beginning of the 1990's integration into international markets and technological improvements pushed the agricultural frontier towards the Espinal and Chaco regions which were traditionally cattle ranching areas of the north of Argentina (González-Roglich et al., 2015; Pengue, 2014). The biodiversity of the Argentine Pampas is notable with 1,600 species of vascular plants (374 grasses), emblematic mammal species such as the "venado de las pampas"(*Ozotoceros bezoarticus*) or the grey pampean fox (*Lycalopex gymnocerus*) and around 500 bird species such as the "ñandú"(*Rhea Americana*) or the chaja (*Chauna torquata*) (Herrera et al., 2014).

The Espinal constitutes a transition zone between the Pampas and the Chaco and is characterized by shrublands (mainly "calden" (*Prosopis caldenia*), "atamisque" (*Caprris atamisquea*) and "pichana" (*Psila spartoides*)) and grasslands, as well as *Prosopis sp., Acacia sp.*, and *Aspidosperma sp.* trees (Burkart et al., 1999; Guida Johnson and Zuleta, 2013). The Espinal can be divided into two regions based on district boundaries and the proximity of ecological characteristics (Figure I-2) and thus can be assigned to either the Pampas or the Chaco.

Land-use changes in Northern Argentina and associated environmental impacts

Argentina is the second largest country in Latin America covering over 2.7 million km^2. Due to its recent rapid agricultural expansion (Baumann et al., 2016), Argentina is one of the world's leading agricultural producers and major exporter of agricultural commodities (Leguizamón, 2016; Pengue, 2014).

Although Argentina increased its agricultural production at the beginning of the 1950's (Waters et al., 2016), it was not until twenty years later, with the incursion of soybean feed and the use of pesticides and fertilizer inputs, that agricultural production sky rocketed. For example, while in 1970 Argentina produced only about 26 tons of soybean feed, this yearly output more than doubled by 2010 to 53 million tons per year (Leguizamón, 2014). Production increases, particularly of soybean, maize, and wheat, have translated into the massive expansion of cropland into Argentina's forests, especially in the Chaco region of Argentina's north (Aide et al., 2013; Baumann et al., 2016; Hansen et al., 2013). Similarly, cropland, especially for soybean feed, is increasingly replacing pastures in the Argentine Pampas region, and in turn leading to the relocation of less-profitable activities, such as grazing, to frontier regions such as the Chaco, further increasing conversion pressure of forest to grazing land. The production increases came at high environmental costs for example, the depletion of a number of important ecosystem services (Baumann et al., 2016; Macchi et al., 2013; Mastrangelo and Laterra, 2015; Torres et al., 2014).

To date, only a few studies have assessed the interrelationship of the forces of land-use change in Argentina but these contain three major gaps. First, only spatial determinants such as soil quality, or climate were used (Gasparri et al., 2015), thereby missing the underlying causes of land-use changes. Second, only one of the two eco-regions were assessed (Bert et al., 2011; Choumert and Phélinas, 2015; Zak et al., 2008), thereby neglecting potential connections between them. Third, and finally, only forest loss was studied (Gasparri et al., 2015; Volante et al., 2016), thereby potentially missing the interactions between multiple land-use change. Therefore, an integrated analysis of causes and spatial determinants of land-use/cover dynamics in Argentina that incorporates spatial determinants and underlying causes of change that focuses in the two most productive agricultural regions of Argentina is essential for informing sustainable land-use planning.

Potential future pathways in Northern Argentina

With increasing population numbers, changes in diets and climate, Argentina may remain one of the world's leading producers and exporters of agricultural commodities (Ministerio de Agricultura Ganadería y Pesca Argentino, 2011). Argentina has also been on the leading side of technological developments with a very high use of genetically modified organisms (GMOs) since the early development of agriculture (ISAAA, 2016; Leguizamón, 2014).

Yet there is still ample scope for the development of more mechanized practices and increasing the efficiency of crop yields, by the usage of fertilizers, or on meat per cow produced by increasing the fattening cycle of calves (Forte, 2016). Argentina has the capacity to expand its agricultural production by increasing the intensity of the production but there are still also vast areas of semi-natural forests that can accommodate new agricultural activities (Baumann et al., 2016; Ramankutty et al., 2002). All this brings important uncertainties on whether and where future land-use changes may occur and the extent, magnitude and consequences that these changes may have on the remaining ecosystems and their functioning. Therefore, understanding the possible effects that potential future land-use changes can have on the environment is an important issue for land-use planning in Argentina in order to avoid unwanted outcomes from management and decision making.

To date there are few exercises that develop scenarios in order to assess the effects of planning decisions or specific policies in Argentina. Of those that have been developed, they, i) quantify the amount of land that would be deforested or occupied by agricultural practices in qualitative ways (Adamoli et al., 2011; Pengue, 2014), ii) describe the increase in production under different productive options (Canosa et al., 2013), iii) are used as a decision support system (Recatalá Boix and Zinck, 2008), iv) infer potential land-use changes based on environmental conditions (Tittonell et al., 2006), or v) qualitatively describe storylines for the country (Patrouilleau et al., 2015; Patrouilleau et al., 2012). However these studies do not spatially allocate where these changes may occur, nor do they integrate the assessment of potential outcomes of development policies in the most agriculturally productive regions of Argentina - which is of high importance for spatial and conservation planning.

2.2 Research Questions and Objectives

In the context of agricultural conversions that affect natural ecosystems in South-America, the goal of this thesis was to ***understand the drivers of agricultural land-use change to explore future land-use changes and how economic and conservation policies may influence these changes in Northern Argentina.***

This goal can be translated into three overarching research questions that will be addressed in this thesis:

<u>Research Question I:</u> *What drives agricultural expansion and intensification in Argentina?*

Understanding the underlying causes and spatial determinants of land-use change and to what extent they play a role in Argentine land-use conversions is of high importance to inform decision making for the sustainable planning of the country. Additionally, studying the conversions among multiple different agricultural land uses instead of only the conversions from forest to agricultural practices enlarges our understanding of landowner decision making in the region and can inform spatial management to plan for potential unwanted results of land-use changes.

The main objectives related to Research Question I were to:

(1) identify underlying causes of land-use change in the dynamic agricultural frontiers of Argentina, the Pampas , the Espinal and the Chaco regions, between 2000 and 2010 and,
(2) identify the spatial determinants of land-use change in the dynamic agricultural frontiers of Argentina between 2000 and 2010

<u>*Research Question II:*</u> *How may land use change under different economic and conservation policy scenarios?*

Gaining knowledge on the likelihood but also the location of potential future land-use conversions and associated deforestation builds resilience when planning for conservation. Moreover, the assessment of potential future effects of economic or conservation policies on the spatial distribution of land-uses and/or on the maintenance of semi-natural ecosystems is of high value for both spatial and conservation planning in Argentina.

Research Question II required two research objectives, each related to one policy arena in Argentina. The specific research objectives were to:

(1) assess potential spatial effects of different future economic policies in the dynamic agricultural frontiers of Argentina: the Pampas, the Espinal and the Chaco regions, between 2010 and 2030, and
(2) evaluate the potential influence of different future potential conservation policies on the connectivity of the dry forest of the Argentine Chaco.

<u>*Research Question III:*</u> *What are potential environmental impacts of future spatial policies in Argentina?*

Assessing the impact that policies can have on the environment is the goal of conservation and spatial planning in order to adapt management decisions and steer landscapes into desired directions. An integrated landscape management, where socio-economic developments are included in conservation strategies, can be beneficial to plan future land conversions and minimize environmental impacts on priority areas for conservation. The specific research objectives were to:

(1) detect areas of conservation concern where future agriculture developments may impact important areas for conservation , and
(2) quantify the impacts in terms of natural ecosystem cover loss.

2.3 Approach and structure of this thesis

Answering the main research questions in this thesis for Argentina, although highly relevant and interesting, poses several challenges due to the areal extent and population size of the country, the cyclical economic crises and political instability experienced, and the lack of consistent data gathering, storage and coherent data distribution and availability. The research presented in this thesis, however, circumvents these difficulties by gathering a coherent set of economic, environmental, social and planning data to build a consistent database. This in-depth database is crucial in developing statistical models and scenarios that gain a deep understanding of underlying and spatial determinants of land-use change in the Pampas, the Espinal and in the Chaco regions of Argentina for the years 2000 and 2010 and the potential implications of spatial policies in the region.

This thesis is structured in three core chapters (chapter II, III and IV) that develop each of the three research questions above and two more chapters that introduce (chapter I) and synthesize the main findings of chapters II-IV (chapter V) (Figure I-3). Chapter II, III and IV were each written as a stand-alone scientific article and were either published or submitted to international peer review journals as follows:

Chapter II: *Piquer-Rodríguez, M., Butsic, V., Gärtner, P., Macchi, L., Baumann, M., Gavier-Pizarro, G., Volante, J., Gasparri, I., Kuemmerle, T. (in review) Drivers of agricultural conversion in Argentina 2000-2010. Environmental Research Letters.*

Chapter III: *Piquer-Rodríguez, M., Baumann, M., Butsic, V., Gasparri, I., Gavier Pizarro, G., Volante, J., Müller, D., Kuemmerle, T. (in review) The potential impact of economic policies in future land-use conversions in Argentina. Journal of Land Use Policy.*

Chapter IV: *Piquer-Rodríguez, M., Torella, S., Gavier-Pizarro, G., Volante, J., Somma, D., Ginzburg, R., Kuemmerle, T. (2015) Effects of past and future land conversions on forest connectivity in the Argentine Chaco. Landscape Ecology 30(5):817-833*

Chapter II addresses research question I by gathering a coherent set of economic, environmental, social and planning data to build a consistent database of spatial factors that drive agricultural land-use change. This database was used to construct a net returns model that enables the selection of the most relevant underlying causes and spatial determinants of agricultural land-use change in Argentina between 2000 and 2010. This chapter built the basis for future simulations under different economic policies (Figure I-3).

Chapter III and IV address research question II and III by investigating potential land-use conversions from economic and conservation policies (research question II) and evaluating environmental effects from these policies (research question III). Chapter III builds on the previous net returns model (chapter II) to simulate plausible scenarios based on potential economic policies and assesses areas of conservation concern. Chapter IV develops a spatial simulation model of potential deforestation and applies a range of potential land-use zoning options based in the Argentine Forest Law (Law 26331). Following this, it evaluates landscape connectivity under several conservation strategies (Figure I-3).

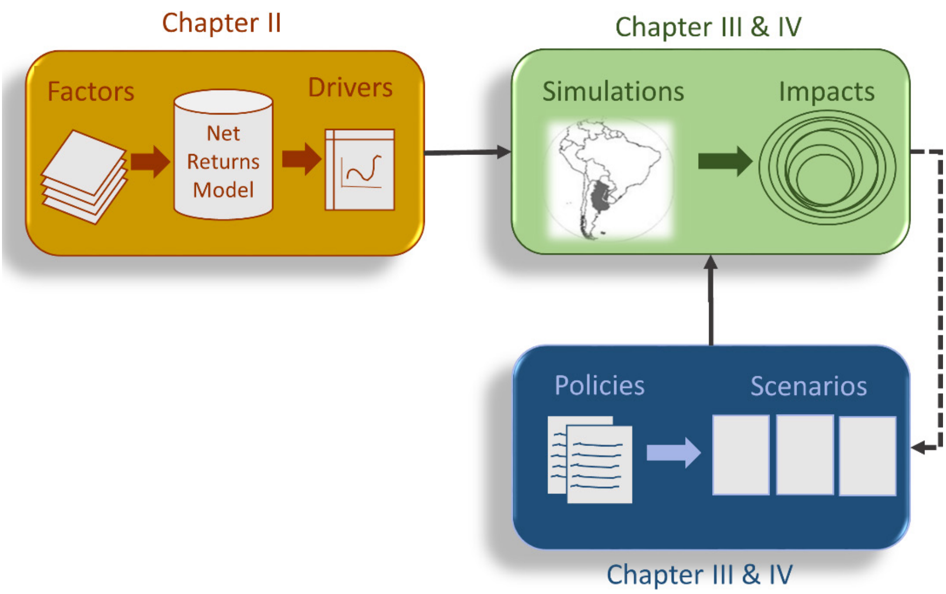

Figure I-3: Schematic overview of the conceptual framework of this thesis.

Chapter II:
Drivers of agricultural land-use change in Argentina

Environmental Research Letters.
Special Focus Issue Tropical Dry Forest Ecosystems and Ecosystem Services in the Face of Global Change (in review)

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Abstract

Agricultural expansion and intensification in South America's dry forests and grasslands increase agricultural production, but also result in major environmental trade-offs. The Pampas and Chaco regions of Argentina have been a global hotspot of agricultural land-use change since the 2000s, yet our understanding of what causes these land-use changes, and what influences their spatial patterns remains partial. We parameterized a net returns model of agricultural land-use change to estimate the probability of agricultural expansion (conversions of woodlands to either cropland or grazing land) and agricultural intensification (conversion of grazing land to cropland). Uniquely, this allowed us to jointly quantify the importance of a range of underlying causes and spatial determinants, for multiple agricultural land-use changes, for Argentina's prime agricultural regions as a whole. We found that cropland expansion occurred mainly in areas of better agro-environmental conditions, whereas grazing land expansion occurred mainly in areas less suitable for cropping. Yet, cropland and grazing land expansion in the Chaco were overall much less sensitive to profit-related factors than agricultural intensification in both regions. Profits were a particularly strong cause of intensification in the Pampas, where cropland profits rose by 29% (compared to 18% in the Chaco). This suggest that further agricultural intensification into Argentina's remaining natural areas, such as the Chaco, is likely as long as agricultural demand and returns to agriculture continue to be high. However, non-market features, such as zoning, tenure insecurity, or cultural ties to the land, seem to be important to explain land-use/cover changes in some regions of the Chaco. Overall, the moderate impact of profit-related factors on mediating woodland conversion suggests economic policies (e.g., taxes or subsidies) are unlikely to alter conversion rates and patterns dramatically. Zoning may be a more powerful tool for governing land-system dynamics.

1 Introduction

Humans have transformed the Earth for millennia by converting natural areas to agriculture (Foley et al., 2011). These conversions have resulted in a substantial increase in food production, but have also led to decreasing biodiversity, increasing carbon emissions and diminishing non-provisioning ecosystem services (Gibbs et al., 2010b; Newbold et al., 2015; West et al., 2010). Such trade-offs are especially important in the world's tropical dry forest and savannas, which harbor high biodiversity and other natural resources, yet are under intense land-conversion pressure (Aide et al., 2013; Laurance et al., 2014; Portillo-Quintero et al., 2015). Understanding the underlying causes of land-use change in these regions, as well as the factors determining the location of these changes, is important to inform land-use planning and to avoid unwanted outcomes (Foley et al., 2005; Tilman et al., 2011).

The decisions of individual farmers, agricultural enterprises, and governments to expand or intensify agriculture are taken locally, but depend on a range of underlying causes operating across multiple scales (Angus et al., 2009; Reidsma et al., 2006). At the global scale, factors such as human population growth (Godfray, 2011; Hazell and Wood, 2008; Tscharntke et al., 2012), changing diets (Bajzelj et al., 2014; Lambin and Meyfroidt, 2011; Tilman et al., 2011), as well as bioenergy production influence the demand for agricultural products and thus, international commodity prices (Geist et al., 2006; Lambin et al., 2003). Likewise, at the regional scale, the level of technological development, political instability, or cultural ties to the land can play important roles in how land-use decisions are taken (Gasparri and le Polain de Waroux, 2015; Thomas et al., 2014). At even finer scales, a range of spatial determinants such as soil quality, climatic patterns, or socio-economic characteristics influence where agricultural expansion and intensification take place (Golub and Hertel, 2008; Lambin et al., 2013; Lubowski et al., 2008; Meyfroidt, 2015). To identify policies that can influence land-use change toward desired outcomes, it is important to understand the relative significance of different underlying causes, and how they interact with spatial determinants describing variable site-conditions (Levers et al., 2014; Meyfroidt, 2015).

Spatial Net Return Models (NRM) are powerful tools to assess the combined impacts of underlying causes of land-use change, such as agricultural profitability, while controlling for spatial determinants influencing the configuration of land use (Bockstael, 1996; Butsic et al., 2011). The basic intuition of these models is that individual land owners maximize the utility from land use (Capozza and Helsley, 1989). In cases where land is used primarily as an input to production, utility can be proxied well by economic net returns (i.e., profit or loss). This theoretical concept can be translated to a statistical model via regression techniques (Wooldridge, 2011), allowing to model observed land-use change in terms of economic (including non-market) rents (Irwin and Bockstael, 2004; Lewis et al., 2009).

South America has been, and will likely continue to be, a global hotspot of agricultural expansion and intensification, with major trade-offs in terms of biodiversity and no-provisioning ecosystem services (Aide et al., 2013; Laurance et al., 2014; Ramankutty et al., 2002). Within South-America, dry forests and grasslands are particularly prone to land-use change, especially in Brazil, Paraguay, Bolivia, and Argentina. The Pampas grasslands and Chaco dry forests of Argentina (Baldi and Paruelo, 2008; Gasparri and Grau, 2009; Grau et al., 2015; Volante et al., 2016) have experienced especially high increases in agricultural production, particularly of soybean since the end of the 1990's (Baumann et al., 2016; Pengue, 2014), bolstering Argentina's role as a world-leading agricultural producer. At the same time, this has triggered land-use changes at unprecedented rates, resulting in the widespread loss and fragmentation of natural vegetation (Adamoli et al., 2011; Aide et al., 2013; Piquer-Rodríguez et al., 2015; Viglizzo et al., 2010). Similarly, cropland is increasingly replacing grazing land in both, the Argentine Pampas and Chaco regions (Gavier-Pizarro et al., 2012; Lende, 2015).

Despite these rapid land-use changes, few studies have assessed the causes of agricultural expansion and intensification in Argentina, and these suffer from one or more of the following shortcomings. First, existing studies have focused only on spatial determinants such as soil quality or climate (Gasparri et al.), thereby neglecting the underlying causes of land-use/cover changes. Second, existing work has typically focused on small regions, typically inside a single ecoregion (Bert et al., 2011; Choumert and Phélinas, 2015; Zak et al., 2008), thereby neglecting potential connections between ecoregions. Third, those studies that have assessed underlying causes have neglected the location factors determining land-use/cover change patterns (Bert et al., 2011), thereby disregarding the substantial spatial heterogeneity that exists across Argentina. Finally, existing work has typically only studied forest loss (Gasparri et al., 2015; Volante et al., 2016), thereby potentially missing the interactions between multiple agricultural expansion and intensification.

This translates into a substantial knowledge gap in our understanding of what drives land-system dynamics in some of the world's prime agricultural regions. To address this research gap, we developed a spatial net returns model of land-use/cover change between the years 2000 and 2010 to understand land-use dynamics in the Argentine Pampas and Chaco ecoregions. Our approach is, to the best of our knowledge, novel in that we (a) jointly model agricultural expansion and intensification, (b) assess both underlying causes and spatial determinants of land use/cover dynamics, (c) model multiple land-use/cover changes simultaneously, and (d) assess all of Northern Argentina's major agricultural regions. Specifically, we addressed two main research questions:

1. How did the underlying causes related to agricultural profitability affect land-use/cover change in the Pampas and Chaco regions between 2000 and 2010?
2. Which spatial determinants influenced agricultural land-use/cover change patterns in the Pampas and Chaco regions in that period?

2 Material and methods

2.1 Study Area

Our study area encompassed the main agricultural regions of Argentina: the Pampas, the Espinal and the Chaco ecoregions (~1.3 million km^2, Figure II-1). Soy accounts for half the grain production in Argentina and more than half of the cropped area in the country (Lende, 2015). Cattle ranching is also widespread with approximated 3 million tons of meat produced per year (www.minagri.gob.ar/ganaderia), of which 10% is exported (Santarcángelo and Fal, 2009).

The Pampas region encompasses the provinces of Buenos Aires, Entre Rios, Santa Fe, southern Cordoba, La Pampa and San Luis (Figure II-1). The region is of flat terrain, subtropical to temperate climate (mean annual temperature is ~15 °C, with an average monthly maximum of 22 °C and annual precipitation ranges 1100mm in the east to 800mm in the west) (Cabrera, 1971) and soils very rich in organic matter (Paruelo et al., 2007). Its natural vegetation are grasslands, mainly composed of *Stipa sp*., *Briza sp*., *Bromus sp*., and *Poa sp*. (Cabrera, 1971). The Chaco region extends into the provinces of Formosa, Salta, Jujuy, Chaco, Corrientes, Santiago del Estero, northern Cordoba and Santa Fe, Catamarca and Tucuman, and generally is characterized by flat terrain, except for the west and southwest were terrain is rougher, and a semi-arid and highly seasonal climate (mean annual temperature is ~22 °C, with an average monthly maximum of 28 °C and annual precipitation ranges 1200mm in the east to 450mm in the west) (Morello et al. 2012). The Chaco is characterized by tree species of the genera Schinopsis and Aspidosperma ("quebrachos") (Prado, 1993). The Espinal constitutes a transition zone between the Pampas and the Chaco and is characterized by tree species such as *Prosopis sp.*, *Acacia sp.*, and *Aspidosperma sp.*, shrubs and grasses (Burkart et al., 1999). For our study, we distributed the Espinal among the two other ecoregions based on proximity and ecological characteristics of districts (Figure II-1).

2.2 Data used to characterize underlying causes and spatial determinants of land-use change

We generated land-use/cover maps for the years 2000 and 2010 based on existing maps for cropland (Volante et al., 2015) and forest cover (Hansen et al., 2013), and considered all other lands that were neither water, urban areas, nor had slopes of more than 5 degrees as in principal suitable for grazing (Volante et al., 2015). Based on overlaying these maps, we mapped conversions of grazing land to cropland, woodland to cropland, and woodland to grazing land between the years 2000 and 2010. These land-use/cover changes formed the dependent variables in our model. The accuracy of our land-use/cover change maps, evaluated using independent data was 90%. See Text SI II-1 for a detailed description of the land-use/cover change map and the accuracy assessment. Our independent variables comprised economic factors, mainly variables related to cropland and grazing land profits at the district level (i.e*., departamentos*) in 2010, as well as spatial determinants of agricultural land-use change at the 1-km gridcell level, mainly climatic (i.e., aridity), accessibility (i.e., travel distance to provincial capitals), topographic (i.e., slope), edaphic (i.e., soil productivity) and cropland neighborhood variables (i.e., neighbors and area share in 2000) (Table II-1). See Text SI II-2 and Text SI II-3 for a detailed description of the independent variables.

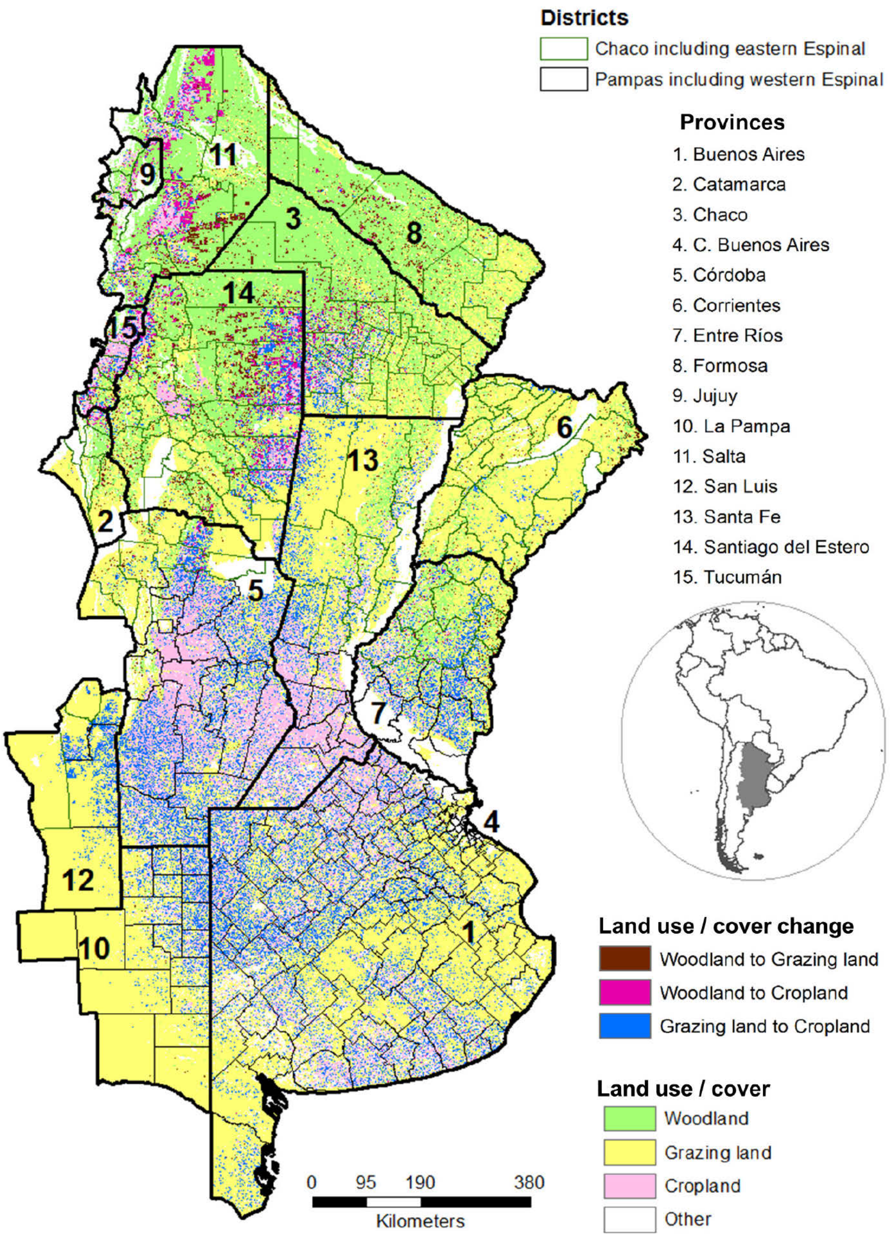

Figure II-1: Agricultural land-use and land-cover changes between 2000-2010 in the Chaco and Pampas region (including the transitional Espinal ecoregion) that we studied.

2.3 Statistical approach

We modeled three agricultural land-use/cover changes for our study region using two net returns models. First we used a logit model to assess the conversion of grazing land to cropland in the Chaco and Pampas, and second we used a multinomial logit model to assess the conversion of woodland to cropland and woodland to grazing land in the Chaco. We parameterized the multinomial model for the Chaco region only, because the few forests in the Pampas mainly represent commercial forest plantations, and forest loss there thus represents forest management, not land-use change. Using this probabilistic framework, we estimated the likelihood of a parcel of land (represented by a grid-cell of 1x1 km^2) converting from woodland to either grazing land or cropland, or converting from grazing land to cropland.

The key for estimating a net returns model is to calculate the returns of the current land use alternative to all possible land uses. Because our maps did not distinguish between different crop types, we calculated average district level net returns for crops (Lubowski et al., 2008). We used data on the national average price for main crop types (e.g., "*pricecrop*" in USD/ton (t)), district data on crop yields (e.g., *"yieldcrop"*, t/ha), the percentage of agricultural land in a district in a given crop (e.g., *"%crop"),* and the cost to produce each crop summarized at the ecoregion level (e.g., *"costcrop",* USD/ha). We then calculated the average profit to cropland per ha, per district for all crops under study (i) (i.e., soy, sorghum, sunflower, corn, wheat and cotton) as (Equation (II-1)):

$$\text{Average profit to cropland } = \sum_{all\ crops}(\%crop_i) * \left((yieldcrop_i * pricecrop_i) - costcrop_i\right) \quad \text{(II-1)}$$

To calculate average profits to grazing land per district, we used district-level live meat yield data (*yieldmeat,* t/ha) and multiplied this by the national internal producer price (*pricemeat*, in USD/t) and subtracted the direct costs of production at the ecoregion scale (USD/ha, see Appendix for detail). We adjusted the profits to account for the costs of deforestation. We did this by dividing the cost of 200USD/ha by 15 (years) and then subtracting this annual cost from the annual profits.

Using these profit calculations, along with our independent variables, we estimated the probability that a parcel of land would convert from grazing to crop as (Equation (II-2)):

$$Y^* = B_0 + B_1 province + B_2 * 1cropneighbor2000 + B_3 * cropneighbor2000 + B_4 * cropprofit + B_{5-8} * soil + B_9 * chaco + B_{10} * grazeprofit + B_{11} * slope + B_{12} * distcapital + B_{13} * \%crop2000 + B_{14} * aridity + B_{15-22} * Interactions + e_i \quad \text{(II-2)}$$

Where Y^* is the latent variable and the error term is distributed with a standard logistic distribution, e~Logistic (0,1). *Cropprofit* and *grazeprofit* represented profits for each use at the district level (as in equation II-2).

Table II-1: Description of variables used to parameterize the net returns model.

Variables	Description	Units	Spatial Resolution	Sources
Land-use/cover Conversions				
Grazing land to Cropland	Conversions from grazing land to cropland	0-1	1km^2	Volante et al. 2015, own data
Woodland to Grazing land	Conversions from woodland to grazing land	0-1	1km^2	Hansen et al. 2013, own data
Woodland to Cropland	Conversions from woodland to cropland	0-1	1km^2	Hansen et al. 2013, Volante et al. 2015
Environmental				
Aridity	PP/PEVT in 2010	-	1km^2	INTA weather stations
Soil	FAO's index of soil agricultural productivity	0-4	1km^2	Atlas de suelos, INTA
Slope	Degrees of slope	degree	1km^2	www.landcover.org (SRTM)
Economic				
pricecrop	Producer prices at the first point of sale	USD /t (current $)	Country	FAO stats
pricemeat	Live meat price	USD/t (current $)	Country	FAO stats
Yieldcrop, yield meat	Crop yields meat produced	t/ha	Department	Databases Integrated System of Agricultural Information (SIIA in Spanish) and Stock cattle INTA2010
Costcrop, costmeat	Direct costs for crop and meat production	USD/ha (current $)	Ecoregion	INTA, Margenes Agropecuarios, MAyG
Distcapitals	Cost distance to provincial capitals using roads in 2010	USD (current $)	1km^2	IGN-SIG250
Structural				
Protected Areas	Network of Protected Areas	0-1	Country	World Database on Protected Areas, www.wdpa.org
Provinces	Control variable (dummy)	character	Province	Database of Global Administrative Areas
Ecoregion	Control variable (dummy)	1,2	Ecoregion	WWF
%cropland2000	Crop area per department in 2000	ha	Department	self-generated
1cropneighbor2000	None or >=1 crop neighbors in 2000	0,1	1km^2	self-generated
cropneighbor2000	Number of cropland neighbors in 2000	0-8	1km^2	self-generated

We used neighbor variables in 2000 (i.e., *1cropneighbor2000, cropneighbor2000)* to characterize cropland in 2000 because they account for unobserved characteristics such as existing infrastructure, technical developments, access to capital or land ownership patterns. Expansion next to existing cropland was characterized by the variable *%crop2000* (see Text SI II-3 for more details). We further interacted *cropneighbor2000* with *crop/grazeprofit* and *soil* for both cropland and grazing land to account for variation in the impact of net returns given the factors that influence yield, soil, the ecoregions and the number of neighbors. In case of the logit model, we additionally interacted with the *chaco* dummy variable. An identical set of covariates was used to estimate a multinomial logit model.

The resulting regressions contained more than 30 independent variables, including the interactions, making the interpretation of model coefficients complex. To facilitate the interpretation of our modelling results, we calculated marginal effects of each variable on the predicted probabilities of conversion, and plotted predicted margins (i.e., probabilities of conversions) across the distributions of our suite of variables, holding all other variables at their mean. The interactions between variables were fully accounted for in these simulations and standard errors were estimated using the delta method (Oehlert, 1992; Williams, 2012). See Text SI II-4 for more detail on the model specifications and interpretation.

2.4 Comparison of actual and predicted land-use change

One major assumption of our models was that land users will maximize the economic profitability of land. However, in reality a number of factors may prevent this from happening such as indigenous cultural ties to the land, or existing land-use zonation that prohibit certain conversions (i.e., Argentina's Forest Law, which designates some woodlands where certain land conversions are not allowed). To explore how far actual agricultural land-use changes deviated from those predicted by our net returns model, we summarized the actual land-use/cover change data from our maps at the district level, and compared it to the average predicted conversion probabilities at the district level in 2010.

3 Results

In 2000, about 22% of our study area was woodland (of which 96% was located in the Chaco), 54% was grazing land and 16% was cropland (Figure II-2). In the Chaco, woodlands decreased from 40% of the region in 2000 to 36% in 2010. This is equivalent to an annual forest loss of 0.4 percentage points or ~3,400 km², a number almost three times higher than the global annual deforestation rate between 2005 and 2010 (0.14 percentage points) (FAO, 2012). About 40% of all woodland loss was due to conversion from woodland to cropland, while 60% of the change was due to woodland conversion to grazing land. About 19% of the grazing land loss was due to conversion to cropland that happened at some point between 2000 and 2010. Overall, cropland expanded to 23% of the landscape in 2010, almost a doubling compared to 2000 (Figure II-2).

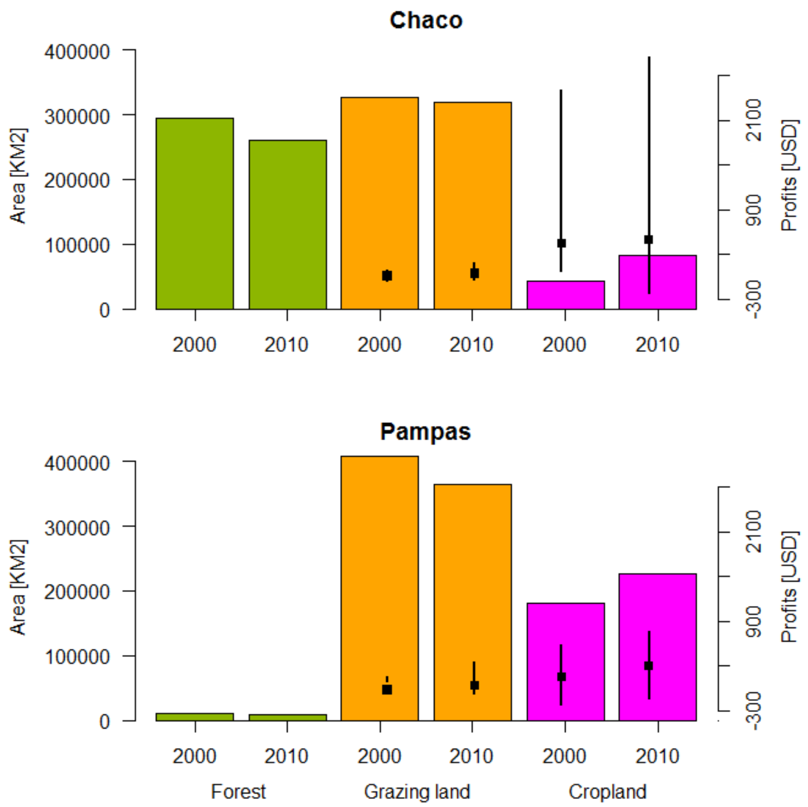

Figure II-2: Extent (km^2) of land-use /cover classes for the studied period (bar plot) and agricultural profits (USD) represented by the mean (dot) and min/max (length of solid vertical lines).

Among the variables that characterized land-use/cover change, crop neighbor effects and environmental characteristics were the strongest determinants of land-use/cover change in both ecoregions (Table II-2). *Aridity* positively influenced the likelihood for woodland to grazing land conversions (for a one-unit increase in aridity, the probability that land-use/cover converts increased by 0.12, i.e., 12 percentage points), but had a negative bearing on the conversions to cropland (decreasing 16 and 7 percentage points when converting from woodland and grazing lands respectively; Table II-2). Conversions to croplands were more likely in areas of higher soil productivity (increasing 14 percentage points when converting grazing land and 5 percentage points when converting woodland), whereas conversions to grazing land was more likely on soils with lower productivity. Likewise, increasing soil productivity increased the likelihood of conversion from grazing land to cropland over ten percentage points for class two (medium) and four (very high) compared to class zero (no productivity), while increasing from class zero to class three increased the likelihood of conversion by over 14 percentage points (Table II-2). Increasing *slope* or *distcapitals* decreased the likelihood of conversions to both cropland and grazing land (e.g., decreasing between 2 to 5 percentage points when increasing slope by 1 degree). However, *distcapitals* was not statistically significant for conversions from woodland to cropland (Table II-2). Although increasing *crop/graze profits* increased the likelihood of woodland-to-cropland and woodland-to-grazing conversions, this was not a significant cause of land-cover change in the Chaco (Table II-2). In the Pampas, *cropprofit* was significant and positively influenced conversions to cropland.

Table II-2: Logit and multinomial logit regression model estimates with marginal effects (coeff). Statistical significance of $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$***. Profits coefficients are related to the end state of the land-use conversion (e.g., for conversions to cropland, profits coefficients are based on cropland profit).

Variables	**Grazing land to Cropland**		**Woodland to Cropland**		**Woodland to Grazing land**	
Environmental	*coeff*	*p-value*	*coeff*	*p-value*	*coeff*	*p-value*
Aridity	-0,0701	0.008**	-0,1665	0,212	0,12257	0.008**
Slope	-0,0470	0***	-0,0222	0.008**	-0,03079	0.003**
Soil low	0,0455	0***	0,0246	0.001**	0,00965	0,19
Soil medium	0,1000	0***	0,0414	0***	0,00552	0,64
Soil high	0,1438	0***	0,0384	0.007**	0,02967	0.014*
Soil very high	0,1013	0***	0,0502	0***	0,01952	0.02*
Economic						
Distcapitals	-0,0007	0,109	-0,0003	0,698	-0,00085	0.019*
Cropland / Grazing profits	0,0001	0.002*	0,0000	0,499	0,00035	0,15
Spatial						
cropneighbor2000	0,0273	0***	0,0062	0.001**	-0,00673	0,136
%crop2000	0,1425	0***	0,2476	0***	0,06052	0,186
1cropneighbor2000	0,0863	0***	0,0742	0***	0,02603	0.002**
Chaco	-0,0934	0.001**	-	-	-	-
Regression models	**Logistic**		**Multinomial logistic**			
	Numb. Obs.	334597	Numb. Obs.	133400		
	Pseudo R^2	0,25	Pseudo R^2	0,11		

The coefficients of *cropneighbor2000* were significant and positive for conversions to cropland. For example, increasing from zero to eight cropland neighbors in 2000 (*cropneighbor2000*) increased the likelihood of conversion from grazing land to cropland by 30 percentage points (calculated as coeff *1cropeighbor2000* +8*coeff *cropneighbor2000;* Table II-2). *1cropneighbor2000* is a factor variable and thus the increase in the likelihood of conversion from grazing land to cropland when changing from zero to 1 or more neighbors in cropland in 2000, was 8.6 percentage points (Table II-2). Grid-cells, either in woodland or grazing land, which had at least one neighboring grid-cell in cropland (*1cropeighbor2000)* had approximately 8% higher likelihood to convert to croplands than those which had no cropland neighbors, when holding all variables at their means (Table II-2).

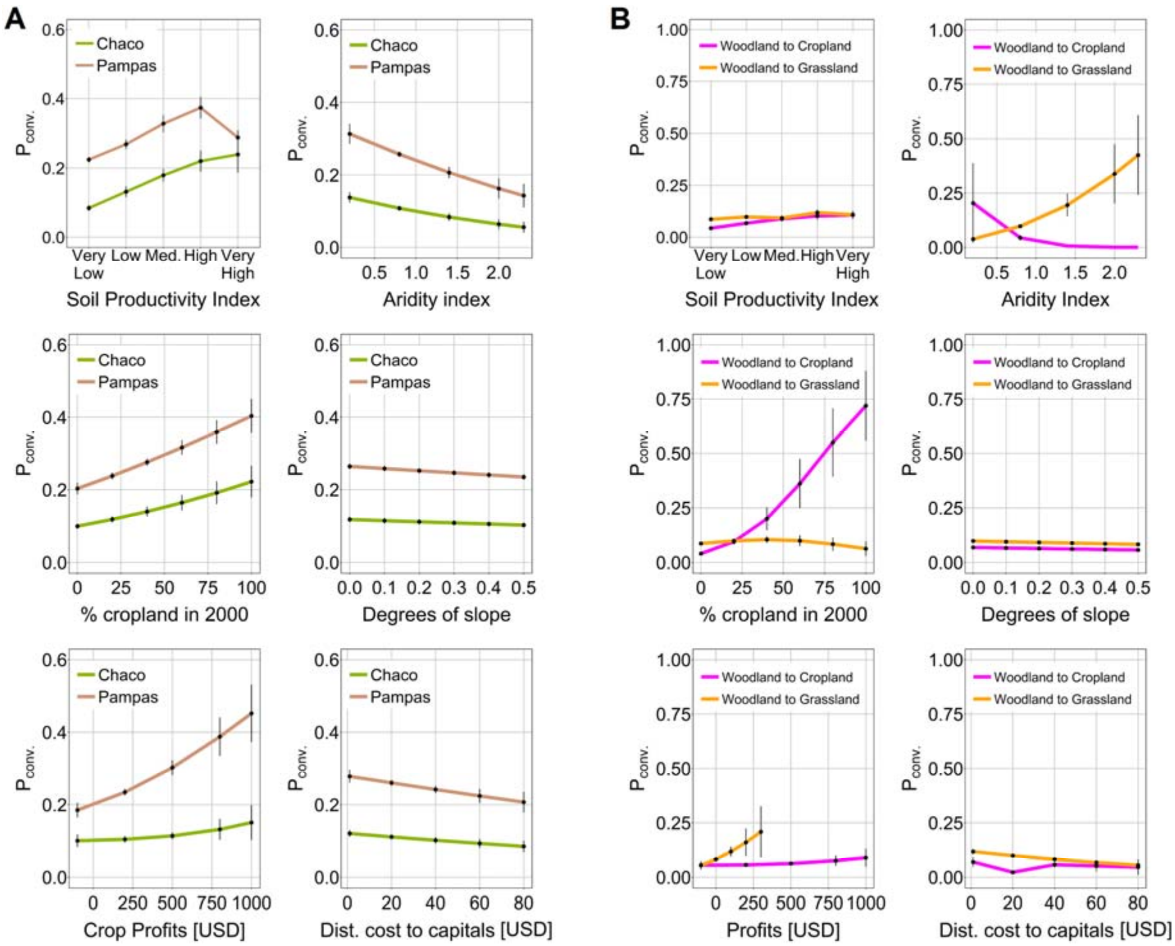

Figure II-3: Predicted probabilities (Pconv) of land-use/cover conversions for selected variables across their distribution, while holding all other variables at their means. (A) Conversions from grazing lands to croplands in the Chaco and the Pampas. (B) Conversions from woodland to cropland and grazing lands in the Chaco. Profits shown are those after conversions took place (i.e., for conversions from woodland to cropland, the probabilities are plotted based on changes in cropland profit).

The proportion of cropland in 2000 at the department level (*%crop2000*) increased the likelihood of conversions in general and was always positively and significantly correlated with the likelihood of conversion to cropland. Grid-cells that were located in districts with higher cropland proportion in 2000 (*%crop2000*) had 14% and 25% higher likelihood of conversions from grazing land and woodland to cropland respectively, when holding all variables at their means (Table II-2).

The predicted margins also showed that increasing profits increased the probabilities of conversions from grazing land to cropland in both regions. However, land in the Pampas was both more likely to transition to cropland, and more sensitive to increases in cropland profits than land in the Chaco (Figure II-3A). *Slope*, *soil* productivity, *%crop2000* and *crop/graze profits* had similar impacts in each model (Figure II-3). However, *aridity* had different impacts for conversions to grazing land or cropland, where conversions to grazing land were more likely under more arid conditions than conversions to cropland (Figure II-3). *Distcapitals* showed also an interesting break in the decreasing probabilities of conversions from woodland to cropland at intermediate costs (Figure II-3B).

Comparing the actual and simulated land-use/cover change at the district level for 2000-2010 showed generally high concordance, especially for the Pampas region (Figure II-4A). However, conversions from grazing land to cropland in the Chaco (Figure II-4A), showed high probabilities of land-use conversions for some districts that actually had fairly low conversion rates in 2000-2010, especially in the provinces of Salta and Santiago del Estero. The opposite was the case for conversions from woodland in some districts in the provinces of Chaco, Salta and Cordoba, where actual conversions rates were high but we predicted comparatively low probabilities of conversions (Figure II-4B and Figure II-4C).

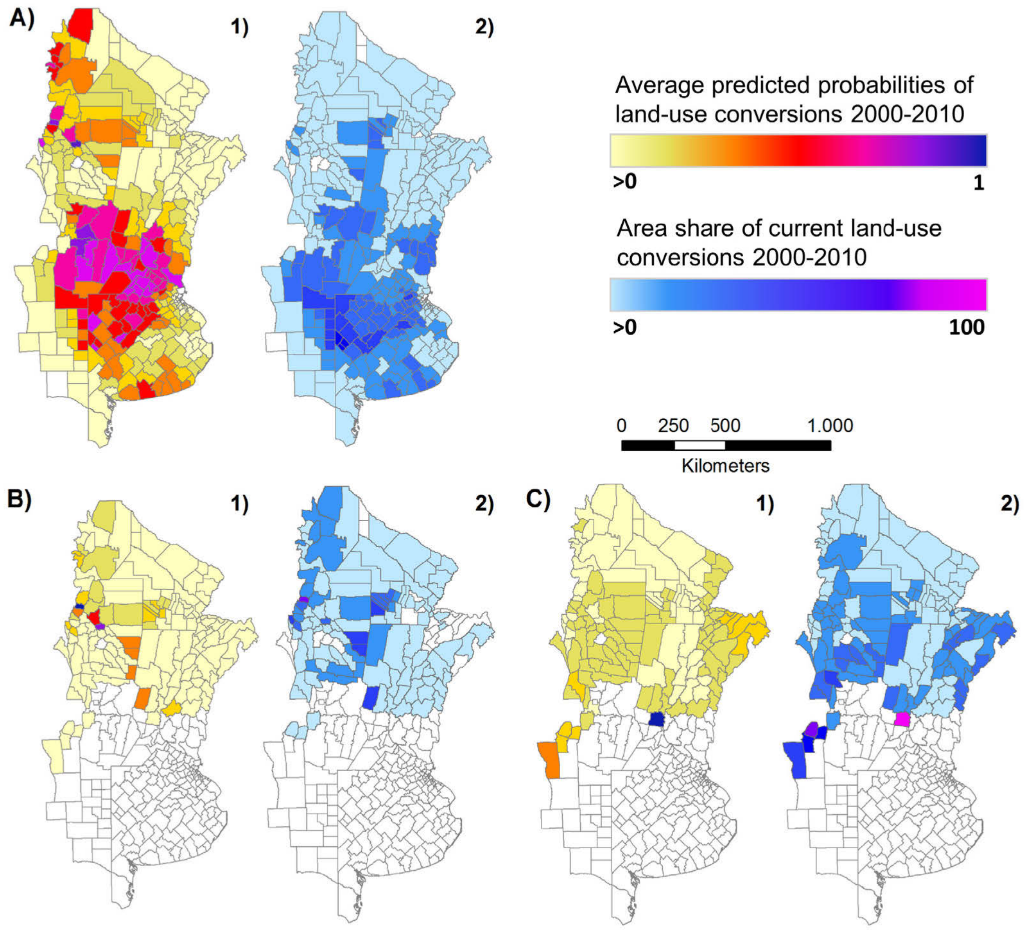

Figure II-4: Predicted (1) and actual (2) land-use/cover conversions at the district level for (A) grazing land to cropland, (B) woodland to cropland and (C) woodland to grazing land. The concordance of these two maps (i.e., districts with high average predicted probability of conversion and a high share of actual land conversions in 2000-2010) point at districts where land-use decisions are captured well by the factors entailed in our net returns model. Where these two maps disagree, factors other than those entailed in our model may have affected land conversion rates

4 Discussion

Many subtropical and tropical dry forests and grasslands are currently undergoing widespread agricultural expansion and intensification, especially in South America. While these land-use changes lead to an increased provisioning of agricultural commodities, they come at the cost of substantial decrease in terms of non-provisioning services and biodiversity. Understanding what drives spatial patterns of agricultural land-use change in these systems is therefore important in order to identify policies that allow mitigating these tradeoffs and steering land-use dynamics towards desired pathways (Aide et al., 2013; Hill and Southworth, 2016; Kuemmerle et al., 2016). Here, we address this research gap by jointly modelling, to our knowledge for the first time, the dynamics of agricultural expansion and intensification for two of the most important agricultural regions of

Argentina, the Pampas and the Chaco regions, for the time period 2000 and 2010. To do so, we developed a novel, spatial net returns model that allows to assess both the importance of a range of underlying causes and spatial determinants of agricultural land-use/cover change.

Our analyses provide a number of key insights. First, agricultural expansion patterns were closely related to specific environmental and structural spatial determinants, suggesting that cropland mainly expanded into areas of better agro-environmental conditions, and ranching into areas less suitable for cropping. This fits well with land rent theory (Lambin, 2012), considering the comparatively higher returns from cropping compared to grazing, and is observed elsewhere in South-America (de Espindola et al., 2012; Müller et al., 2011). Second, we found that agricultural expansion into dry forests was less sensitive to profit changes than agricultural intensification (i.e., conversions from grazing land to cropping), mainly because it is nearly always profitable to convert woodlands to croplands or grazing land. Third, the Pampas, that has a longer cropping history, seemed overall more responsive to changes in the variables that we modelled, and thus to marginal profit changes, than the Chaco. Finally, comparing actual and predicted patterns of agricultural land-use/cover change showed that the economic factors were important in some regions (especially in the Pampas), but factors not included in our economic model, such as zoning, non-market features, or cultural ties to the land seem to be important to explain observed land-use/cover changes in parts of the Chaco.

4.1 Underlying causes and spatial determinants of agricultural land-use/cover change in the Pampas and Chaco

Conversions from woodland to cropland were influenced more by environmental spatial determinants than conversions from woodland to grazing land, whereas transportation costs were not a major determinant of cropland expansion. This likely reflects the economies of scale that exist in the region (i.e., increasing agricultural production resulting in a proportionate saving of transport costs due to expanding infrastructure). Profits are likely less dependent on local logistics and infrastructures, since agribusiness producers own their export facilities and infrastructure (Lende, 2015). This is also apparent in Figure II-3B, where the likelihood of cropland expansion into woodlands decreased with increasing cost distance only up until 20 USD, but not thereafter. This points at a potential cost threshold that may be a constraint for small producers that can amortize transportation investments less than large producers (Sanchez et al., 2007). The relatively low importance of profits in determining woodland to cropland conversions can also be explained due to the preference of some land-users for expanding in isolated areas, where lower governmental control on land acquisition and rights exists (Leake and Economo, 2008).

Conversions from woodland to grazing land systems were mainly characterized by the environmental suitability of a location. Grazing mainly expanded into more arid areas, likely because silvopastural and intensified pasture systems are better adapted to conditions of higher aridity and are more resilient to water scarcity than cropping systems (Houspanossian et al., 2016). Moreover, higher soil suitability led to lower likelihood of grazing land expansion, possibly because grazing is still possible even in areas with lower soil quality that is too poor to crop (Demarco, 2010). Cost distances to provincial capitals lowered the likelihood of conversions to grazing land, indicating that transportation costs play a role in expanding agricultural frontiers driven by cattle ranching (Gasparri et al., 2015). Overall, as in the case of cropland expansion, the expansion of grazing land into woodlands was relatively insensitive to grazing profits. The purposeful stagnation of cattle productivity of the early 2000's by farmers aiming at increasing the value of their cattle stock (Santarcángelo and Fal, 2009) may have strongly influenced the low sensitivity of grazing expansion to profits.

In contrast to agricultural expansion, agricultural intensification (in our case the conversions from grazing land to cropping) was highly sensitive to profit-related variables and occurred mainly in areas characterized by lower aridity and high soil productivity. This is consistent with land rent theory (Lambin, 2012), suggesting that the decision to crop or graze on existing agricultural land may indeed be motivated by changes in profit at the margin of these land uses (i.e., the additional profit that motivates the investment). In other words, if there are small changes in grazing relative to cropland profits, we would expect large changes in land use, because these systems are both highly profitable, and land-use actors may thus be less capital-constrained in terms of shifting from one land use to another. Our neighbor cropland variables further highlighted the cropland agglomeration taking place in the Pampas, due to knowledge and technology transfers, similar as elsewhere in South America (Garrett et al., 2013). Moreover, the longer cropping history of the Pampas (Pengue, 2014) may further explain its higher responsiveness to drivers of agricultural change than the Chaco. However, the fact that the Chaco is less responsive to drivers of agricultural change than the Pampas does not necessarily rule out that agricultural intensification may not take place in the future in the Chaco where this process is already happening.

4.2 Comparing observed and predicted land-use/cover change

Comparing predicted and actual land conversions showed that some districts had higher probabilities of land-use/cover conversions than were actually observed between 2000 and 2010, especially in the provinces of Salta and Santiago del Estero (Figure II-4). Under the current national zoning plan (i.e., Forest Law, implemented in 2007; Figure II-4 and Figure SI II-3), much of these districts fell into the “sustainable use” zones (i.e., yellow zones, where full deforestation is not allowed), or “no use’ zones (i.e., red zones, where agriculture is excluded). This suggests that zoning was, at least to some extent, successful in steering forest loss away from these zones, as observed for Argentina (Nolte et al., 2017a). A second factor explaining lower than expected conversion rates are indigenous communities that manage forests communally and whose livelihoods depend on non-timber forest products, such as in some areas of Salta province (http://www.mapaeducativo.edu.ar/pueblos_indigenas/).

Conversely, some districts had higher-than-expected woodland conversion rates, especially in the provinces of Chaco, southern Salta and northern Cordoba. Many of the areas where this was the case were zoned as “productive” zones (i.e., green zones where deforestation is allowed) (Figure II-4B) and deforestation there may take place due to fear of future change in zoning, even if the current land utility is not high. Interestingly, areas of highest agreement of actual and simulated conversions from woodlands to grazing lands were mostly located in the Espinal, where historically expansion of grazing land towards marginal lands from the Pampas took place (Pengue, 2014), or in ‘yellow’ zones, where conversions to silvopastures (pastures with trees) are allowed in some provinces (Figure II-4C and Figure SI II-3).

4.3 Limitations

Our model, and hence our results, are not without limitations. First, high quality agricultural production and land-use/cover data for a mid-point in our study would have allowed us to more precisely estimate the impact of changing returns for intermediate land uses (e.g., woodlands converting into grazing lands before a second conversion to croplands), zoning policies and landscape configurations on land conversion, but such data was not available. Second, our model assumed landowners maximize profit and thus disregards landowners that have a preference for cultivating the land in traditional ways, not focusing on maximizing profitability. Likewise, our model does not capture well land users’ decisions who are mainly interested in securing land rights or claiming land as a commodity for future speculation (Leguizamón, 2016). However, given the advanced agro-business setting that represents Argentina's agriculture sector today, our assumptions are reasonable for most areas of our study region, and discrepancies between predicted and observed land-use/cover change are informative to understand where other factors are at play. Third, our model is trained during a time frame of strong economic turmoil in Argentina. At this time, land owners had less capacity to react to economic incentives then they would in more stable times. To some extent then, actors may have been limited by economic constraints (such as access to capital) that are not explicitly modeled here.

5 Implications and Conclusions

Overall, our study highlights that agricultural intensification processes in the Pampas and Chaco ecoregions were more sensitive to marginal profit changes than the expansion of agriculture into woodlands, as highlighted by the lower importance of profit-related factors for these conversions (Table II-2) and the sometimes high discrepancy of observed and predicted land-use/cover change for the Chaco (Figure II-4). Thus, as long as global demand for agricultural products keeps growing and returns to agriculture also remain high, both of which is very plausible, continued agricultural intensification in Argentina's Pampas and Chaco regions is likely. These finding translate into a number of important messages in the context of land-use and conservation planning and policy making. First, policy interventions that target profits, such as raising taxes, providing subsidies or implementing payments for ecosystem services programs are potentially powerful in promoting or inhibiting land-use intensification, especially in the Pampas, but less likely to slow down or even curb deforestation in the Chaco. Second, forest loss did not happen in some regions where our economic model suggests conversion pressure should be high (Figure II-4). While further analyses are needed to better understand what led to the preservation of these forests, our findings highlight that other factors are at play and that land-use zoning (such as implemented via the Forest Law or protected areas) and community-based management (as is the case for some indigenous lands) can influence land-use/cover change patterns in the Argentine Chaco in major ways, possibly resulting in less forest conversion (Nolte et al., 2017a). This highlights the power of zoning to protect natural ecosystems and their services in agricultural frontiers, and the opportunities that lie in the upcoming revision of the Forest Law in steering land conversions away from ecologically sensitive areas. More generally, our study shows how spatial models of net returns can improve understanding of land-use drivers in areas undergoing rapid land-use change.

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Supplementary Information

Text SI II-1: Dependent variables: data on agricultural land-use change.

Generation of land use/cover change map - An accurate land use/cover change map spanning the entire study region was not available at the spatial and thematic detail required for our study. We therefore generated such a map for the period 2000-2010 at a resolution of 1x1 km² using multiple, existing datasets, and we used an independent set of ground-truth points to rigorously validate the resulting maps. We focused on four stable classes (*woodlands*, *croplands*, *grazing lands*, and *other*) as well as three transition classes (woodlands to grazing lands, woodlands to croplands, grazing lands to croplands).

We obtained cropland information from two existing cropland maps generated from MODIS imagery at a spatial resolution of 250x250 m^2 for the years 2000 and 2010 with an overall accuracy of around 80% (Volante et al., 2015). The cropland areas in the map considered annual crops that were cultivated in highly managed agricultural systems (i.e., winter, summer, double, or irrigation cropping). Due to the large extent of our study region, to minimize computation time, and because some of our predictors were coarser-resolution, we aggregated the cropland maps to a 1x1 km² resolution using a nearest neighbor resampling method (Figure SI II-1).

We derived information of woodlands from a high-resolution map of changes in tree/woody vegetation cover based on Landsat TM and ETM+ images at a spatial resolution of 30x30 m^2 (Hansen et al., 2013). This dataset contains fractional woody vegetation for 2000 and woody cover loss for 2010, and we used a threshold of 25% to separate woodland from non-woodland areas. We derived the woodland cover in 2010 by subtracting the woodland cover in 2000 minus the woodland loss in 2010. We aggregated the resulting binary map to 1x1 km² using a nearest neighbor resampling method (Figure SI II-1).

The “other” class contained permanent water bodies (such as rivers, ponds or lakes), salt plains without vegetation, urban areas and areas with slopes of more than 5 degrees (Volante et al., 2015). Towns of more than 50,000 inhabitants where manually digitized, whereas towns of more than 1,000, but less than 50,000 inhabitants where assigned an entire 1x1 km² grid-cell (Instituto Geografico Nacional, 2015).

We merged the three layers woodland, cropland and “other” for both time points (2000 and 2010) into a single land-use/cover map, by hierarchically overlaying them. First, we integrated the woodland and the cropland layer, by giving precedence to cropland areas where cropland and woodland areas overlapped. We then overlaid the ‘other’ layer on top of the cropland/woodland-map, thereby giving precedence to the ‘other’ class. All other areas were considered grazing areas. This is a reasonable assumption for our study region in Argentina, since virtually all non-cropland areas containing some kind of vegetation are being grazed, even if they are inundated for parts of the year (Baumann et al., 2016). Thus, according to that definition, grazing lands contained natural grasslands, savannas, and pastures, both with native herbaceous vegetation as well as implanted grasses (Figure SI II-2). We considered the “other” class as stable, meaning that none of the ‘other’ classes in either time period could transition. After generating the two individual land use/cover maps for 2000 and 2010, we applied map comparison to derive our change classes of interest for the period 2000-2010 (Figure II-1, main manuscript).

Accuracy assessment- To assess the robustness of our land use/cover maps, we carried out an independent accuracy assessment for each of the two land-use maps individually (i.e., 2000 and 2010). To do so, we first generated a stratified random sample of 50 points per land-use class (overall 200 points) per map. We then visually examined each of these points individually based on Landsat image composites (Hansen et al., 2013), and, where available, high-resolution imagery in Google Earth. We then generated the error matrix, and calculated the overall accuracy and class-wise user’s and producer’s accuracies (Foody, 2002, 2008). We also corrected for possible sampling bias (Olofsson et al., 2014). The accuracy of both maps was high, reaching accuracies around 90% (2010 and 2000). The class-wise accuracies were high as well, with the only exception of the “other” class for the year 2000. However, and most importantly, our three target classes (i.e., woodlands, croplands, grazing lands) showed high user’s (average of 92% for the year 2000 map, and 91% for the year 2010 map) and producer’s accuracies (average of 91% for the year 2000 map and 88% for the year 2010 map,Table SI II-1).

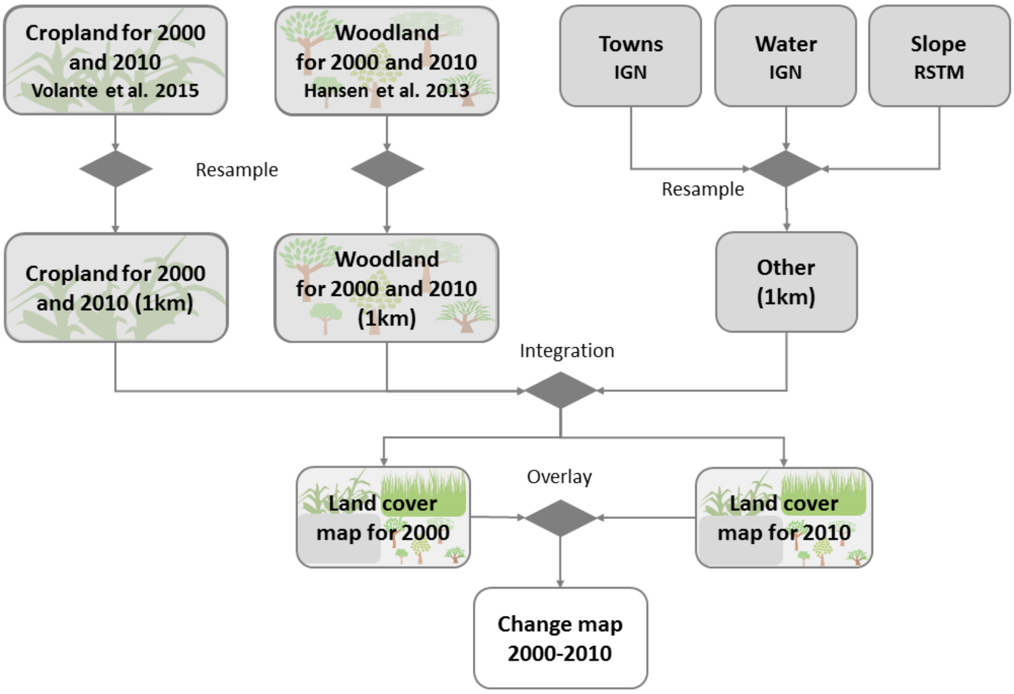

Figure SI II-1: Workflow of the land-use data preparation for the years 2000 and 2010 used in the net returns model: Cropland, Woodland, Grazing land and "Other".

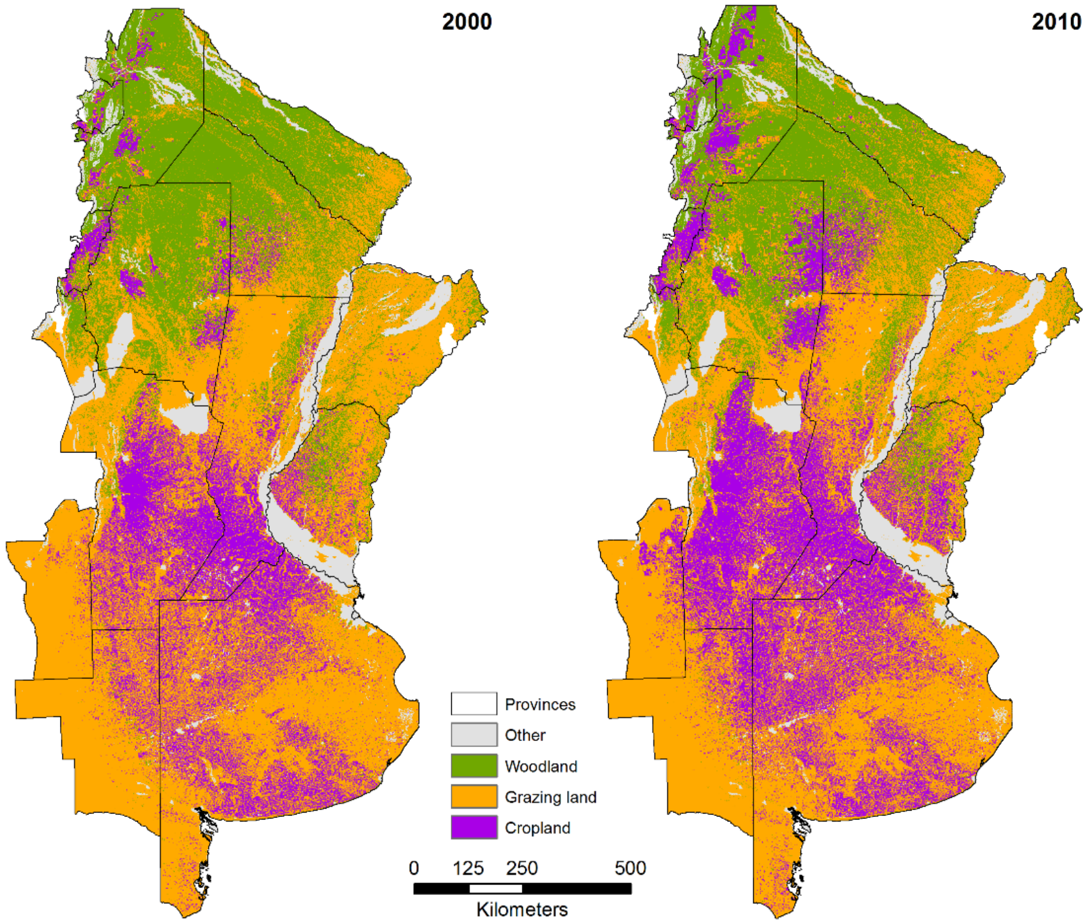

Figure SI II-2: Land use maps for 2000 and 2010 covering the study area.

Table SI II-1: Accuracy assessment results for the land use/cover maps for 2000 and 2010, including the bias/corrected overall accuracy, as well as class-wise user's and producer's accuracies (UA, PA).

Year	**Overall Accuracy (%)**	**Kappa**	**Class Name**	**Class-wise Accuracy** PA (%)	**UA (%)**
2000	90.3%	0.9	Other	67.4%	88.0%
			Woodland	89.9%	94.0%
			Grazing land	96.2%	88.0%
			Cropland	87.9%	94.0%
2010	89.9%	0.9	Other	100.00%	90.0%
			Woodland	88.8%	88.0%
			Grazing land	91.9%	88.0%
			Cropland	84.6%	96.0%

Text SI II-2: Independent variables (I): data on underlying causes of agricultural land-use change

We used multiple data sources to develop a comprehensive database containing information on prices, cost, and production associated with cropping and ranching for our target years 2000 and 2010. Using this database, we calculated cropland and grazing land net returns. First, to assemble a coherent national producer price database, we assumed that all producers were price takers, and were paid the national average local price for their commodities. Since most agricultural commodities produced in our study region are for export, we used internal producer prices for crops and live meat paid at the farm-gate from the Food and Agriculture Organization of the United Nations (http://faostat3.fao.org).

Second, we gathered production costs for crops for 2003 and 2010 from the National Institute of Agriculture Technology (INTA) for the Chaco (Experimental Station Roque Saenz Peña), and Márgenes Agropecuarios for 2000 and 2010 for productive regions in the Pampas (www.margenes.com). We used all direct crop costs, i.e., costs associated with planting (e.g., seeds, work force), crop protection (e.g., pesticides, herbicides, fungicides) and harvesting, and averaged them to the ecoregion level. Direct costs for meat production entailed cattle health maintenance, personnel, and fodder, and were gathered for 2000 and 2010 from the Ministry for Agriculture, Livestock and Fisheries of Argentina (Secretariat of Livestock, http://www.minagri.gob.ar/ganaderia) at the level of productive cattle regions but averaged at the ecoregion level. All costs and prices were obtained in, or converted to, constant US dollars (i.e., USD, adjusted for inflation or devaluation) when necessary.

Finally, we obtained data at the district level on production average yields (t/ha) for main crops (sunflower, sorghum, corn, wheat, soybeans, and cotton, together constituting >90% of all crop production in the study region) from the Integrated System of Farming Information of Argentina (http://www.siia.gov.ar/) for 2002 and 2010. Districts (i.e., *departmentos*) are the smallest administrative unit at which population and agriculture census data are gathered and released consistently in Argentina. A district is equivalent to a county in the USA or the NUTS-3 level in Europe. In cases of no data, we filled gaps with data from 1998 to 2002 for 2000 and 2008 to 2012 for 2010. We also obtained data on the size of the cattle herd and sown area for all crops from the National Agricultural Census of Argentina 2002 (www.indec.gov.ar/Agropecuario) and INTA cattle stock report 2010 (www.rian.inta.gov.ar/ganaderia). We calculated meat yields for each department in t/ha as:

cows * 0.35 [t per cow]/4)/ grazing land [ha] (Nasca et al., 2015).

This assumes that 25% of all cattle are slaughtered annually (ONCCA- Ministerio de Agricultura, Ganaderia y Pesca, http://www.agroindustria.gob.ar).

Our calculation of net returns to cropland and grazing land showed that average net returns (per hectare) from cropland increased by 33% from 2000-2010 (2000=$300, 2010=$400) and net returns (per hectare) from grazing land increased by 300% from 2000-2010 (2000=$-17, 2010=$36, see Figure II-2 in the main manuscript). Cropland and grazing land net returns where influenced by both increases in prices and yields.

Text SI II-3: Independent variables (II): data on spatial determinants of agricultural land-use change

Climatic variables - We derived and tested three bioclimatic variables from the set of weather stations of INTA (http://climayagua.inta.gob.ar): mean temperature and average rainfall (using the R package dismo; www.worldclim.org/bioclim) and an aridity index. The aridity index was calculated using INTA's weather station data annual mean precipitation divided by annual mean evapotranspiration. Because weather stations are point data, we interpolated the weather stations database to obtain a continuous dataset of bioclimatic data using ordinary co-kriging with anisotropy and using elevation data as the covariate.

Accessibility variables - We generated and evaluated five accessibility variables:

(I) Distance to main rivers (i.e., Euclidean distance of every grid-cell to the closest river of the Plata river, Uruguay river, Parana river, Salado del Norte river, Bermejo river, Pilcomayo river or Paraguay river) to proxy accessibility to waterways.

(II) Euclidean distance to paved roads in 2010 to proxy accessibility to main national roads.

(III) Cost distance to towns smaller than 50,000 inhabitants (i.e., cost distance in USD based on the road network in 2010) to proxy local transportation costs. Transportation costs for (III) and (IV) were taken as 0.1 USD per ton and kilometer travelled for paved roads, 0.2 USD for gravel and dirt roads, and 1 USD for land without roads (Müller et al., 2011).

(IV) Cost distance to provincial capitals (i.e., cost distance in USD based on the road network in 2010) to proxy regional transportation costs. Transportation costs used were the same as above.

(V) Transportation costs for exporting produce from provincial capitals to export ports to account for potential economies of scale present in the region. To calculate these costs, we used an 'accessibility catchment' approach (Müller and Munroe, 2005): First, we generated a cost distance surface based on the network infrastructure. Second, we calculated around each provincial capital a 'catchment' area based on equivalent cost distance from each grid-cell to provincial capitals (using the origin-destination cost analysis in ArcGIS). Third, we calculated the distance from each provincial capital to the nearest export port (i.e., Buenos Aires and Rosario).

Finally, we assigned that distance value to all pixels within an accessibility catchment.

Topography - We tested two topography variables. Elevation and slope, that we derived from the aggregated Shuttle Radar Topography Mission (SRTM) elevation model at 1x1-km² resolution (www.landcover.org.).

Population density - We also tested the inclusion of population. We calculated population density (population/km²) from the Argentine population census 2001 and 2010 (www.indec.gov.ar). This variable was not significant and did therefore not enter the final model.

Soil productivity – We obtained a soil productivity index from GeoINTA, Soil Atlas of Argentina (INTA (Instituto Nacional de Tecnologia Agropecuaria), 1990), which ranges from very high to no agricultural potential. This variable is based on FAO's index of soil agricultural suitability, and adapted to the Pampas region. The soil properties considered in this index are macroclimate, drainage, texture, cation exchange, organic matter, effective deepness, salinity, sodium, current and potential erosion (INTA (Instituto Nacional de Tecnologia Agropecuaria), 1990). In this study, we aggregated agricultural production suitability based on the "productivity index" classes from 0 - 4, where 4 was the highest soil productivity for agriculture and 0 equaled unsuitable areas.

Neighborhood - We also included neighborhood configurations in 2000 for crop as variables, as well as the district share of cropland area in 2000, to account for existing infrastructure and knowledge transfer. We calculated the number of grid-cell neighbors to a cropland grid-cell in two ways:

I) As a binary variable with 0 or >= 1 neighbours (i.e., *1cropneighbor2000*).
II) As a variable ranging from 0 to 8 neighbours (i.e., *cropneighbor2000*).

The share of cropland was calculated as the percentage of cropland at the district level in 2000 (i.e., *%crop2000*).

Text SI II-4*:* Model specifications: variable selection, sampling and interpretation.

To minimize the influence of outliers in our model (average net returns to cropland in 2010 were around USD400 but some districts had maximum of US$ 3000), we included only grid-cells where agricultural returns were <USD 1000 per hectare, though this only reduced the dataset by less than 5%. We parametrized the models avoiding collinear predictors so for each of the thematic groups of spatial determinants containing several candidate variables, we ran alternative net returns models using only one variable from each group, and selected the variable that increased model performance (AIC and pseudo R^2) the most. We excluded protected areas from our analysis because we do not expect land-use changes there to be major neither to be driven by rent theory.

To account for potential spatially correlated error terms, which can bias coefficients in logit and multinomial logit models, we sampled only every second grid-cell for model parameterization, thus reducing the potential for correlated errors. In addition, we clustered standard errors at the district level, in order to allow for inter-district correlation between error terms. Running the model with even greater sampling distance (up to 8km between grid-cells) did not change the results.

For the logit and multinomial logit models, the marginal effects for continuous variables can be interpreted as the change in probability of a grid-cell converting from one land use to another, for a one unit change in the independent variable. For example, a marginal effect of cropland net returns of 0.000134 means that for a USD100 increase in cropland profits, conversions to cropland would be 1.34% more likely. The marginal effect of factor variables (i.e., categorical and dummy variables) should be interpreted as the change in conversion probability when a grid-cell changes from the base level (e.g., zero in the case of the soil productivity variable) to the level of interest. *Cropneighbo*r200 and *soil* classes are factor variables and the base levels account for no crop neighbor grid-cells or no soil productivity, respectively.

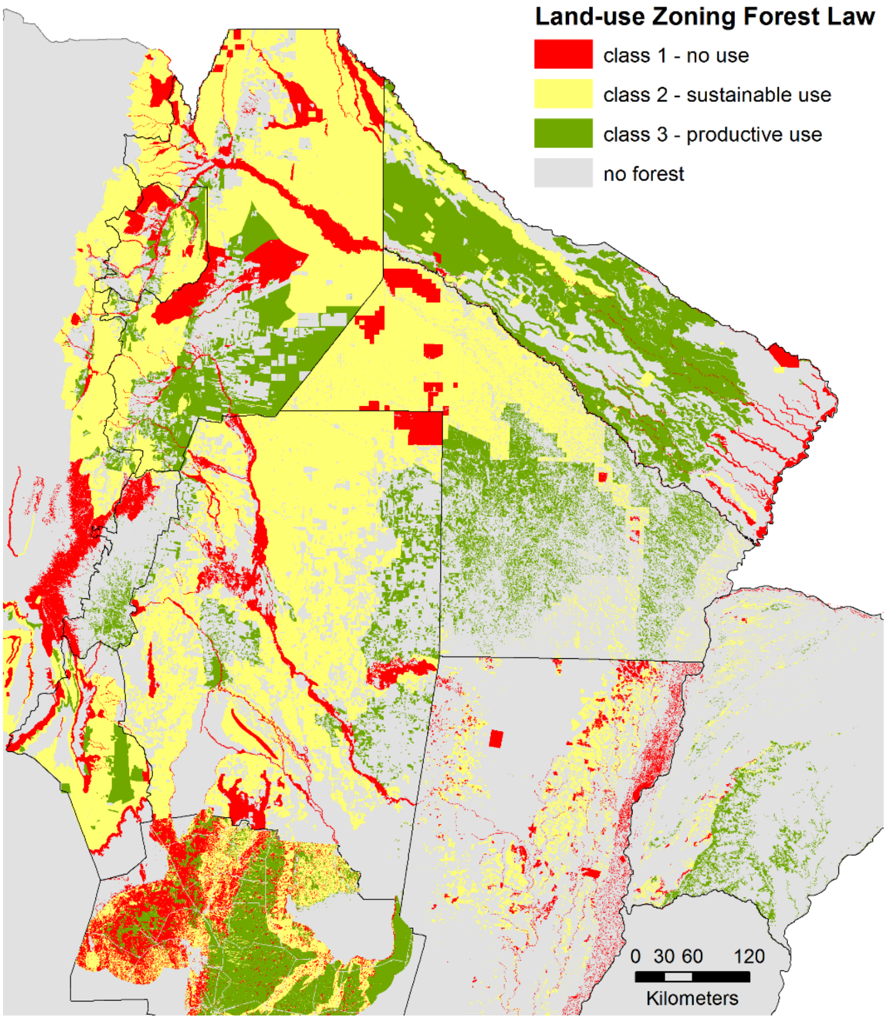

Figure SI II-3: Overview of the land use zonation under the Forest Law of Argentina. Class 3 (green) allows productive uses of the forest, including its conversion to agriculture. Class 2 (yellow) allows sustainable forest uses, and class 1 (red) fully protects forests. Grey background stands for non-forested areas or forests that were not zoned.

Chapter III:
The potential impact of economic policies on future land-use conversions in Argentina

Land Use Policy (in review)

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Abstract

Agricultural expansion and intensification drive the conversion of natural areas worldwide, particularly in the tropics. Scenarios are a powerful tool to explore future land-use trajectories, how these may affect the environment, and how policies may influence them. Focusing on Argentina's prime agricultural areas, the Pampas, Espinal and Chaco, we developed spatially-explicit future land-use scenarios until 2030, considering both agricultural expansion (i.e., conversions from woodland to either grazing land or cropland) and agricultural intensification (i.e., conversions from grazing land to cropland). Our simulations were based on an econometric model of net returns, which assumes profit-maximizing land-use actors, allowing us to assess the amount and spatial patterns of future land-use change. We contrast this with a forecast of future land use based on land-conversion rates from 2000-2010. We systematically test the impact of economic policies (e.g., taxes or subsidies), infrastructure improvement (e.g., road paving), and technological innovation (i.e., yield increases) on land-use conversion rates and spatial patterns. Our results showed that if land users would maximize profits, future land-use change would mainly happen along intensification pathways, whereas deforestation would slow down. This general pattern did not change even for fairly drastic policy interventions. Assuming land-conversions would continue at 2000-2010 rates resulted in continued deforestation, predominantly for cattle ranching in the western Chaco region, and for cropland in both western and eastern Chaco region. Intensification would dominate in the southern Chaco and in the Pampas. Under this scenario, economic policies affected expansion rates of grazing lands in the Chaco markedly, resulting in an agglomeration of cropland, but sometimes also leading to surprising results (e.g., higher deforestation rates for policies reducing profits). Improving the region's road network would create a strong incentive to expand agriculture further into remaining woodlands and grazing lands. Overall, our study provides insights how land-use change in northern Argentina is driven by factors affecting profit margins, but also highlights the importance of other factors (e.g., cultural values, agglomeration factors, actors securing land rights). Thus, our results suggests that economic policies targeting profits (e.g., taxes, subsidies, payment for ecosystem services) may be potentially less powerful in governing future land-use trends than hoped for. Given that our study also highlights the continued high conversion pressure on the region's remaining natural areas, zoning appears to be a more promising and cautionary tool to avoid unwanted environmental impacts in the Chaco.

1 Introduction

Agricultural expansion and intensification drive the conversion of natural ecosystems worldwide, leading to biodiversity loss and the degradation of ecosystem services (Leblois et al., 2017; Maxwell et al., 2016). This is especially the case for the world's tropical and subtropical dry forests, where much of the remaining non-cultivated fertile land is found (Lambin et al., 2013; Laurance et al., 2014; Ramankutty et al., 2002). With ongoing population growth and even faster increasing consumption, the demand for agricultural products is expected to rise dramatically in the 21st century (Foley et al., 2011; Tilman et al., 2011). This will translate into growing pressure to intensify existing agriculture areas, and to expand agriculture into natural ecosystems. Identifying policies that are effective in steering agricultural land-use change, and assessing their relative impact on agricultural expansion versus intensification pathways, is therfore critical (Angelsen, 2010; Meyfroidt et al., 2014).

This requires understanding the underlying forces behind these agricultural land-use changes (e.g., changes in population, diets, market prices) and how they play out given local conditions (e.g., soils, climate, accessability, policies) (Geist and Lambin, 2002; Meyfroidt, 2015). South America harbors some of the world's key agricultural regions, where agricultural land-use change is strongly influenced by global agricultural markets (Byerlee et al., 2014; Gasparri and le Polain de Waroux, 2015). This has resulted in widespread deforestation for cattle ranching and soybean expansion (Baumann et al., 2016; Gasparri et al., 2013; Leblois et al., 2017). Yet, deforestation rates vary starkly from region to region, depending on the environmental characteristics and the national and subnational policy framework (Assunção et al., 2013; Macedo et al., 2012; Nolte et al., 2017b). For example, whereas deforestation rates in the Amazon or the Paraguayan Atlantic Forest have decreased recently (Nepstad et al., 2014; WWF, 2006), in part due to forest protection policies (Baumann et al., 2017; Macedo et al., 2012), agricultural expansion in the neighboring Cerrado and Chaco ecoregions continues unabated (Baumann et al., 2016; Spera et al., 2016). Likewise, agriculture in some regions, such as in the Pampas or the Atlantic Forest, intensifies from cattle ranching to soybean (Bert et al., 2011; Viglizzo et al., 2011; WWF, 2015). In order to efficiently manage the effects of agricultural land-use change, it is therefore crucial to understand its underlying drivers and spatial determinants and to explore the effects of policies on potential future land-use trends.

Scenario analyses are powerful to explore future land-use change and the possible impact of policies on these changes (Gavier-Pizarro et al., 2014; Peterson et al., 2003; Piquer-

Rodríguez et al., 2015; Polasky et al., 2011). Key driving forces that influence land-use change are those directly affecting agricultural profitability assuming that most landowners seek to maximize profits from land use (Bockstael, 1996). Spatial economic models of net returns explicitly model the impact of changes in land profitability (i.e., net returns) on land-use change, while accounting for regional characteristics such as variations in agricultural suitability (Bockstael, 1996; Butsic et al., 2011; Piquer-Rodríguez et al., in review). Once parameterized, such models allow for deep insights into the impact of changes in underlying drivers of land-use change, to explore alternative future scenarios, and to test for the possible effects of specific policies on land-use change (Butsic et al., 2010; Lewis and Plantinga, 2007; Radeloff et al., 2012). This is a major advantage compared to models that project future land-use change based on correlations between past land-use change and its spatial determinants, while typically disregarding the underlying, profit-related causes of land-use change (Plantinga and Lewis, 2014). Yet, to our knowledge, only a few models of net returns have been parameterized for agricultural regions in South America (Arima, 2016; Seo, 2009), and none has used spatial data on agricultural costs and returns to assess profit directly.

Within South America, Argentina is a hotspot of agricultural land-use change, both in terms of agricultural intensification and agricultural expansion (Viglizzo et al., 2011). On the one hand, widespread grazing land to cropland conversion occurs in the Pampas and Chaco ecoregions, mainly for the production of soybean, corn, and wheat. On the other hand, agricultural expansion into the dry forests of the Chaco ecoregion, both for expanding cropland (i.e., soybean, wheat, maize, and cotton) and for expanding cattle ranching are frequent (Baumann et al., 2016; Gasparri et al., 2015; Grau et al., 2015; Volante et al., 2016). These trends have bolstered Argentina's role as a global player in agricultural production and exports since the 1990s (Leguizamón, 2016; Urcola et al., 2015), contributing in major ways to Argentina's economy, and these trends are likely to continue in the future (Laurance et al., 2014; Ramankutty et al., 2002; Schmitz et al., 2014). Yet, these agricultural land-use changes also led to stark environmental trade-offs (Baldi et al., 2006; Baumann et al., 2016; Macchi et al., 2013; Mastrangelo and Gavin, 2014; Torres et al., 2014).

Understanding how policies could influence future agricultural land-use change effectively is therefore a key research field. Policies could target agricultural profits directly, for example via export taxes or through subsidies, as is currently the case (e.g., *retenciones*). More indirect policy measures include agricultural production targets or caps, such as in the Strategic Food and Agricultural Plan (MAGyP, 2011) or the 'Hilton Quota' on beef exports to the European Union (Decree 906/2009 and 1231/2015). Moreover, policies can affect the agricultural sector via infrastructure development (e.g., Infrastructure Investment Plan to 2025 (Bortolín, 2015), Executive Network Framework to 2024 (E.Di.Vi.Ar or Plan Belgrano) via lowering transportation costs, thereby raising land rents (Choumert and Phélinas, 2015). How such policies may influence rates and spatial patterns of future agricultural land-use change in Argentina, however, remains unclear.

Existing work on future agricultural land-use change in Argentina typically explores alternative narratives of potential future agricultural trends (Adamoli et al., 2011; Patrouilleau et al., 2007; Patrouilleau et al., 2012). These studies suggest a growing concentration of land tenure (Bert et al., 2011; Corral et al., 2008), and highlight the potential of intensification for agricultural productivity (Canosa et al., 2013). Because these studies are not spatial, assessing the environmental impact of future land-use and how particular policies would affect these impacts is very challenging though. Conversely, studies that consider the spatial patterns of future land use explicitly were all based on correlative models that are not well-suited for assessing economic policy impacts because they disregard underlying causes of land agricultural conversions, such as land profits (Gasparri et al., 2015; Volante et al., 2016). We know of only one study, from our own prior work, that, spatially and quantitatively, evaluated the impact of zoning on future deforestation in Argentina (Gasparri et al., 2015; Volante et al., 2016), but this study did neither differentiate between different land-use conversions, nor did it include important profit-related causes of land-use change.

Our goal here was to explore potential future pathways of agriculture in Argentina's Pampas, Espinal and Chaco regions. We built on an existing, fine-scaled spatial economic model of net returns, parameterized for the period 2000-2010 (Piquer-Rodríguez et al., 2015). We used this model to analyze potential agricultural expansion and intensification until 2030 and to assess how economic policies may impact these land-use changes, assuming profit-maximizing land-use actors. Given that past land-use changes were likely in part also driven by non-economic factors (e.g., land zonation (Piquer-Rodríguez et al., in review)), we also explore the impact of the same policies on scenarios where we forecast land use based on historical conversion rates. Finally, we compare our future agricultural scenarios with regions of conservation priority to detect possible conflicts. Specifically, we asked three research questions:

1. What are likely rates and locations of agricultural expansion and intensification in Argentina until 2030, assuming profit-maximizing land users?
2. Where would land-use changes occur when forecasting historical (2000-2010) agricultural land-use change rates until 2030?
3. How would different economic policy interventions (e.g., taxes, subsidies, investment into infrastructure) affect land-use change rates and patterns until 2030?

2 Material and methods

2.1 Study Area

Our study area covered the main agricultural ecoregions of Argentina: the Pampas, the Espinal and the Chaco ecoregions (~1.3 million km^2, Figure SI III-1), which are of generally flat terrain, except for some rugged areas in the west. The climate transitions from temperate (Pampas) to subtropical (Chaco), with lower rainfall in the West (800mm) than in the East (1100mm), and the driest parts in the central and southern Chaco (300-400mm) (Herrera et al., 2014; Morello et al., 2012). Soils in the Chaco vary from being rich in minerals and fine in texture in the north (well-suited for agriculture) to the southwest of the ecoregion where soils are sandy with low content in organic matter as in the center of the Espinal (Burkart et al., 1999). The soils in the Pampas are very rich in organic matter (Herrera et al., 2014).

Natural vegetation in the Pampas is characterized by grasslands, mainly composed of *Stipa sp*., *Briza sp*., *Bromus sp*., and *Poa sp.* (Cabrera, 1971). In the Chaco, trees of the genera *Schinopsis* and *Aspidosperma* ("quebrachos") are characteristic, along with Ziziphus ("mistol"), *Prosopis* ("algarrobo"), *Acacia* shrubs and *Cactaceae* in the dry Chaco and *Prosopis* ("algarrobo"), esteppes (Stipa sp.) and palm savannas (*Trithrinax)* in the wet Chaco (Prado, 1993). The Espinal constitutes a transition zone between the Pampas and the Chaco and is characterized by shrublands (mainly "calden" (*Prosopis caldenia*), "atamisque" (*Caprris atamisquea*) and "pichana" (*Psila spartoides*)) and grasslands, as well as *Prosopis sp., Acacia sp.*, and *Aspidosperma sp.* trees (Burkart et al., 1999). Biodiversity in all three regions is high, though protected area networks generally sparse, covering only 1,9% of the total area with many areas of conservation priority outside these reserves (Bilenca and Miñarro, 2002; TNC, 2005). For our study, we split the Espinal ecoregion and merged it to the Pampas or to Chaco ecoregions based on ecological similarity (Figure SI III-1).

The Pampas has a longer land-use history than the Chaco, as cattle ranching has a long tradition in the Pampas due to its flat terrain and productive natural grassland. With the introduction of soybeans in the 1970's, many pastures in the Pampas have been converted into soybean fields and ranching activities were displaced first into the Espinal and then into the more marginal Chaco (González-Roglich et al., 2015; Pengue, 2014). By the end of the 1990s, increasing soybean prices and new genetically modified soybean varieties spurred soybean expansion into the Espinal and Chaco at the cost of native woodlands (Leguizamón, 2016). Between 2000 and 2010, cropland in our study region has expanded steadily by 152,000 km^2 (133,200km^2 from grazing land and 18,300km^2 from woodland) with an additional 27,000 km^2 of grazing land expansion into woodlands. This translated into about 14% of the Argentine Chaco woodlands being converted to agriculture during the 2000's (Baumann et al., 2016).

2.2 Data used for model building

To build our spatial model of net returns (hereafter: Net Returns Model – NRM), we developed a homogenized map of past land-use conversions, pertaining to three types of conversions: (1) grazing land to cropland (here defined as agricultural intensification), (2) woodland to cropland, and (3) woodland to grazing land (here both defined as agricultural expansion). See Text SI III-1 and Piquer-Rodríguez et al. (in review) for further detail. These land conversions formed our dependent variable (Table SI III-1). As predictor variables, we compiled an extensive dataset on crop and cattle yields, internal producer prices, and direct costs in order to generate our cropping- and grazing-related profit variables. We also included a wide range of control variables at 1-km resolution (i.e., pixel size), including climate (e.g., aridity index), accessibility (e.g., travel cost to provincial capitals), topographic (e.g., slope), edaphic (e.g., soil productivity), and neighborhood variables (Table SI III-1).

2.3 Baseline scenarios of future agricultural land use

To assess how land use may change until 2030, we used our NRM to explore land users' decisions of expanding or intensifying agricultural land uses based on productive, economic, and environmental variables for Argentina's prime agricultural regions (Piquer-Rodríguez et al., in review). The NRM was parametrized for the years 2000-2010, using a multinomial logit model for jointly modelling conversions from woodland to either grazing land or cropland, and a logit model for modelling conversions from grazing land to cropland (see Text SI III-1). The econometric estimation of these models is described in detail in Piquer-Rodríguez et al. (in review).

To simulate future agricultural land use, we used the NRM outputs of the likelihood of future land-use conversions for each pixel (Lawler et al., 2014; Radeloff et al., 2012). Generally, for all our simulations (described below), we updated neighborhood variables once in 2020, assuming that environmental conditions, road construction, and profits were static over the time period modelled (OCDE/FAO, 2014). We allowed agricultural land-use changes in accordance with the current zoning policy (i.e., the Argentine Forest Law #26331) that restricts conversions from grazing land or woodland to cropland (Figure SI III 1). Woodland to cropland conversions are only allowed in 'green' zones, woodland to grazing land conversions were allowed in 'green' and 'yellow' zones, assuming that conversions in 'yellow' areas are done under a silvopastoral management plan (i.e., grazing land maintains up to 20% of forest cover, currently *MBGI plans*- Forest Management with integrated cattle ranching) approach . We did not allow for land conversions in currently protected areas ('red' zones).

We simulated four baseline scenarios using our NRM (Table III-1). First, we simulated future agricultural land-use changes assuming that land users seek to maximize profits from land use, as assumed in our NRM (baseline scenario 1 – *BS1;* Table III-1). For our second baseline scenario, we altered BS1 to add increasing crop yields (baseline scenario 2 – *BS2*) through technological innovation and/or improved management, as foreseen by the Strategic Food and Agricultural Plan (MAGyP, 2011). In our third baseline scenario, we assumed historical rates of land-use conversions (i.e., 2000-2010) and constant yields (baseline scenario 3 – *BS3*), thus partly accounting for factors not included in our NRM, such as land speculation, land access and capital availability. Our fourth baseline scenario combined BS3 with the yield increases of baseline scenario BS2 (baseline scenario 4 – *BS4*). See Text SI III-2 for more detail on the four baseline scenarios.

Table III-1: The four baseline scenarios

Baseline Scenario	*Scenario description*
BS1	Land-use change rates from the Net Returns Model (profit maximization) with stable crop yields.
BS2	BS1 with crop yield increases.
BS3	Land-use change rates as in the period 2000-2010 with stable yields.
BS4	BS3 with crop yield increases from BS2.

To model our baseline scenarios *BS1* and *BS2*, we projected the NRM to the years 2020 and 2030 in order to derive land-use transition probabilities. A pixel was assumed to convert from one land-use/cover to another if the simulated probability of conversion was higher than a randomly drawn probability (Radeloff et al., 2012). To ensure model stability and to account for stochastic variability, we ran a Monte Carlo simulation repeating this process 1,000 times, resulting in 1,000 individual land-use simulations for each time step (2020 and 2030). The final land-use class was assigned using a majority rule. To assess the robustness of our simulations, we calculated the deviation of model fit measures (pseudo R^2, AIC) for each simulation (Akaike, 1973; Hu and Palta, 2006), as well as the prediction power by calculating the ratio of observed vs. predicted values from the confusion matrix (Pearce and Ferrier, 2000).

For scenarios *BS3* and *BS4*, we simulated land-use patterns for 2020 and 2030 using our NRM and by forecasting land-use conversions based on the conversion rates observed in 2000-2010 (i.e., assuming constant land-use change). Thus, we assumed 18,300km^2 of woodland to cropland conversions in 2010-2020, and again in 2020-2030. Similarly, we assumed 27,400km^2 of woodland to grazing land conversions and 133,200km^2 of grazing land to cropland conversion for both 2010-2020 and 2020-2030 (Figure III-1). To implement these conversions, we chose the pixels with the highest probabilities of transition based on the NRM simulations until the targeted area for a specific land-use conversion was reached.

2.4 Economic policy interventions

For each of our four baselines scenarios, we evaluated the effect of four contrasting, generic economic policy interventions (PI) on land-use conversions in our study region until 2030. These policy interventions were: (PI-1) decreasing profits from cropping and ranching by 50% assuming raising taxes, establishing new taxes, or raising export caps, (PI-2) increasing profits from cropping and ranching by 50% assuming lowering taxes, installing subsidies, or lowering export caps, (PI-3) improving roads to the provincial capitals, with the goal of connecting these capitals with agricultural frontiers, and (PI-4) improving roads between main towns and major export hubs (including the capitals) , with the goal of connecting these towns to export hubs (e.g., Buenos Aires and Santa Fe's harbors). Together, this resulted in a total of 20 scenarios (i.e., each baseline scenarios without policy interventions, plus four policy interventions; Table III-2 and Text SI III-3).

Table III-2: Description of our four economic policy interventions. Each of these was simulated for the four baseline scenarios

Policy interventions (PI)		*Description*
PI-1	Profit decrease 50%	Implementation of mechanisms that decrease profits for cropping and ranching by 50%.
PI-2	Profit increase 50%	Implementation of mechanisms that increase profits for cropping and ranching by 50%.
PI-3	Road improvement between capitals	Road improvement in the vicinity of provincial capitals.
PI-4	Road improvement around towns	Road improvement in the vicinity of major towns.

Once we simulated future land-use maps for all of our 20 scenarios, we summarized the number of times each pixel was experiencing any agricultural land-use conversion. To highlight where agricultural land-use change may conflict in the future with biodiversity conservation, we compared the map of conversion frequency to the priority areas highlighted in the "*Conservation Portfolio of Priority Areas for Biodiversity"* of the Chaco, developed by The Nature Conservancy (TNC, 2005). *Likewise, we compared our conversion frequency map with* the Valuable Pasture Areas (VPAs) of Argentina, which *are areas of natural grasslands of high conservation value* (Bilenca and Miñarro, 2002). Those TNC or VPA areas that were located within areas of high likelihood of experiencing agricultural conversions were classified as areas of conservation concern.

3 Results

3.1 Future land use in the four baseline scenarios

Simulating future land use assuming stable yields and profit-maximizing actors (*BS1*) showed marked agricultural land-use change between 2010 and 2030. For the first period (2010-2020), we observed a tendency towards agricultural intensification (i.e., grazing land to cropland conversions) whereas the second period (2020-2030) was characterized by both agricultural expansion (woodland to grazing land or cropland conversion) and intensification. Agricultural expansion was not widespread though (2,200 km^2 2010-2020, and 5,800 km^2 in 2020-2030, respectively), resulting in only a moderate (~3%) woodland loss compared to 2010, mainly located in Tucuman, Santiago del Estero and Salta; Figure III-1 and Figure III-2. Agricultural intensification (i.e., grazing land to cropland conversions) covered a staggering 67,500 km^2 in 2010-2020 and 79,600 km^2 in 2020-2030 (Figure III-1). Agricultural intensification occurred more clustered in the provinces of Chaco, Santiago del Estero and Entre Rios and more widespread in Cordoba, Santa Fe, La Pampa and Buenos Aires (Figure III-2). Overall, 96% of the new cropland in 2020-2030 came from agricultural intensification, and only 4% from agricultural expansion. Likewise, 54% of the total deforestation was due to cropland expansion and only 46% was due to grazing land expansion.

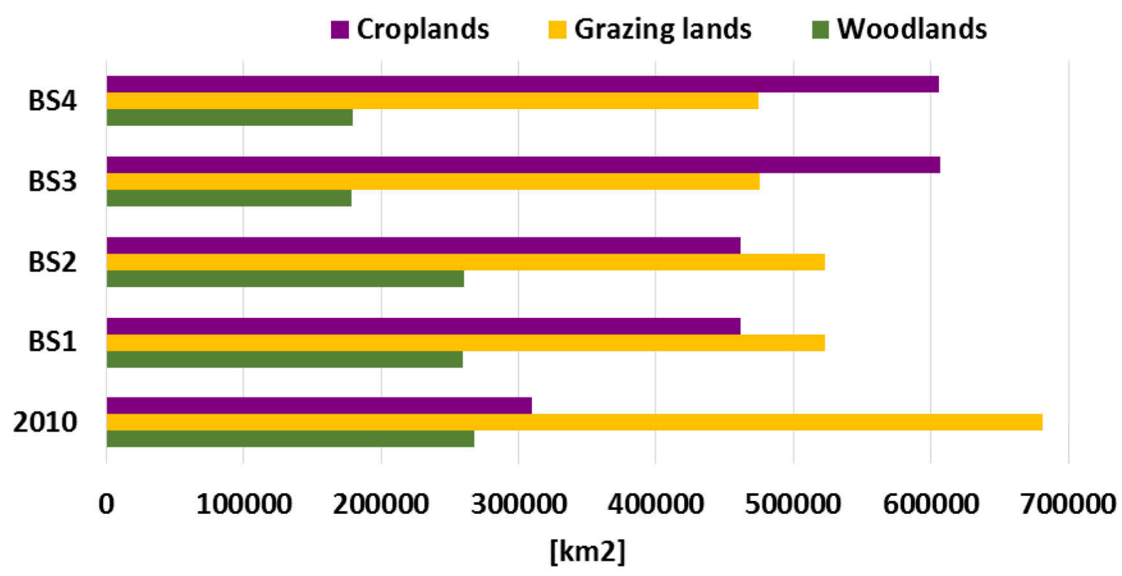

Figure III-1: Area (km2) of land-use/cover simulated for each baseline scenario in 2030. Land use in 2010 is shown for references purposes.

Our second baseline scenario (*BS2*) was similar to the first (*BS1*) but assumed increasing yields. Agricultural expansion and intensification showed similar rates and spatial patterns as in baseline scenario 1 (*BS1*). Agricultural expansion in 2010-2030 was not very widespread, resulting in a similar overall woodland decrease than *BS1* (8,000 km^2). A difference between the two scenarios was that agricultural intensification occurred more spatially concentrated in *BS2* when compared to the *BS1*.

Baseline scenarios 3 (*BS3*) and 4 (*BS4*) assumed future agricultural conversions at rates of 2000-2010. Under both of these baseline scenarios, 33.5% of woodlands in 2010 were lost until 2030 (90,000 km^2), and croplands almost doubled during that period (Figure III-1). *BS3* showed strong cropland expansion in deforestation frontiers, especially in the south of Salta, Tucuman and Chaco provinces (Figure III-2). There was a drastic grazing expansion in Santiago del Estero. Similarly, agricultural intensification occurred clustered in Buenos Aires, La Pampa, Cordoba, Santa Fe and Entre Rios. Assuming yield increase (*BS4*) translated into more concentrated patterns of cropland expansion compared to *BS3,* which translated into woodlands in marginal regions (such as in Chaco or Entre Rios) being spared. Grazing land expansion in Santiago del Estero was even more drastic than in *BS3*.

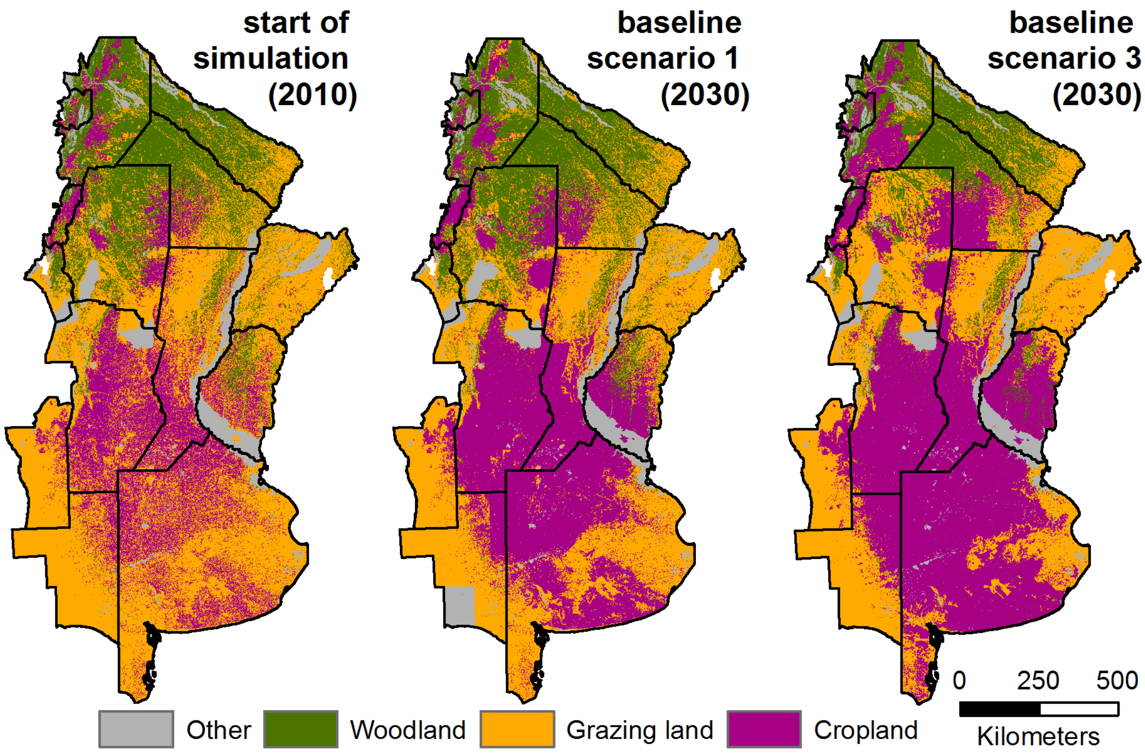

Figure III-2: Future land-use patterns in baseline scenarios 1 (BS1) and 3 (BS3) and start of simulation land use/cover (2010).

3.2 Impact of economic policy interventions on future land use

Comparing future land-use change of our baseline scenarios 1 (*BS1*) and 2 (*BS2*) to those considering policy interventions showed that the impact of these policies on altering future land use was overall fairly limited. Both, policies leading to decreasing (PI-1) or increasing *(PI-2)* agricultural profits resulted in similar overall trends of agricultural expansion and intensification compared to the baseline scenarios. *PI-1* and *PI-2* differed in the spatial patterns of land-use change though, as cropland expansion occurred less clustered under policies that would decrease profits (*PI-1*) compared to the baseline scenarios (*BS1* and *BS2*), whereas under policies that would increase profits (*PI-2*) cropland expansion patterns were more clustered. Paving roads in the future, for enhancing both the connection of provincial capitals or towns (*PI-3 and PI-4)*, translated into a small increase in woodland conversion in marginal regions, such as the case of northern Salta.

Assuming land-use conversions to continue at the rates of 2000-2010 (*BS3* and *BS4;* i.e., higher rates than in BS1 and BS2) increased the impact of our policy interventions. Decreasing cropland and grazing profits by 50% (*PI-1*) resulted into less woodland to cropland conversions in marginal regions of Santiago del Estero and Chaco. Yet, cropland expanded on grazing land in Santa Fe and grazing land expanded on woodland in Santiago del Estero compared to the baseline *BS3* (Figure III-3). Increasing agricultural profits by 50% (*PI-2*) resulted in more agricultural intensification in Santa Fe, while sparing some woodland in marginal regions such as the east of Salta or the north of Santiago. There was also less intensification in southern Buenos Aires compared to the baseline BS3 (Figure III-3).

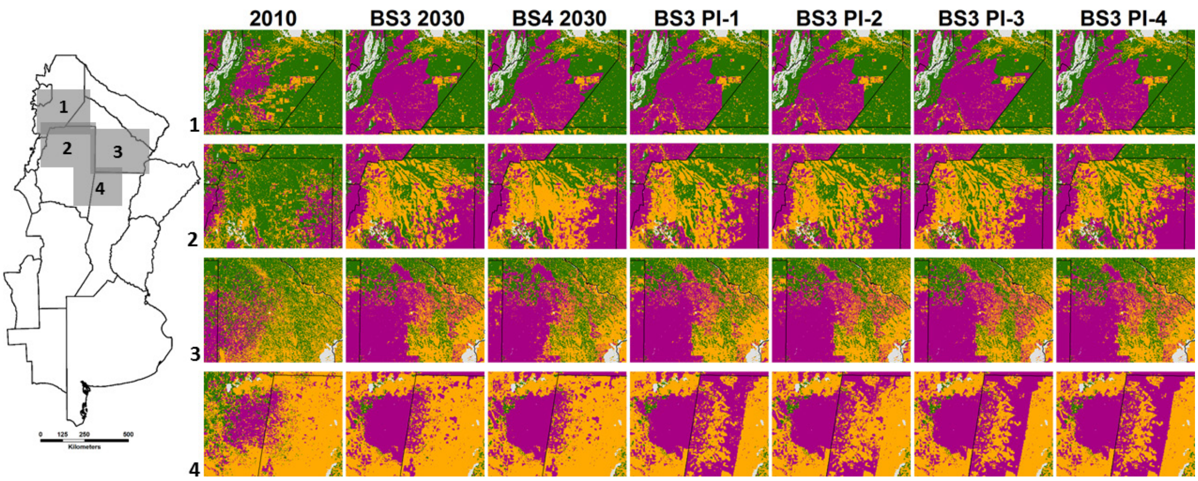

Figure III-3: Detail of spatial economic scenarios in 2030 in the Chaco under the forecast of historical land-use conversions as in 2000-2010. BS: baseline scenario, PI: policy intervention

Road improvements (*PI-3* and *PI-4*) affected land-use change pattern substantially. When focussing on connecting provincial capitals with agricultural frontiers (*PI 3*), grazing land expanded into woodlands in Santiago del Estero, cropland expanded into woodlands in

Salta, and grazing land intensified to cropland in Chaco and in Santa Fe when compared to *BS3* (Figure III-3). Assuming infrastructure investments to better connecting larger towns to export hubs (*PI-4*), we observed less woodland conversion in the east of Salta and the north of Santiago compared to the baseline scenario (*BS3*). Yet, agriculture intensified in Santa Fe compared to BS3 (Figure III-3). Testing these policies when assuming yield increases (*BS4*) showed that there was a general trend towards cropland expansion into woodlands (such as in Entre Rios, Chaco or Salta), but also some sparing effects for woodlands in more marginal regions (such as east of Salta or Chaco).

3.3 Identifying areas with high agricultural conversion pressure

Comparing all our 20 scenarios highlighted some particularly dynamic regions that would experience conversions under most of the scenarios. These regions are primarily located in the south of Salta province (around the town Joaquin V Gonzalez), the south of Chaco province (around the towns of Pampa del Infierno and Charata), the center and north of Santiago del Estero province (around the town Quimili), the west of Santa Fe province (around the town Tostado), the south of Entre Rios (around the town of Villaguay) and the centre and south of Buenos Aires province (around the towns of Chascomus, Rauch, and Olavarria; Figure III-4).

Comparing these areas of high conversion probability to the conservation priority areas highlighted nineteen areas with particularly high land-use pressure (Figure SI III-3, Text SI III-4). Fifteen of these areas belonged to the Chaco (TNC) priority areas (*Transición Chaco-Yungas , Bañados del Quirquincho , Zona del impenetrable , Derrames de los ríos Hornones y Ureña , Bañados del río Salado y Bañados de Figueroa , Bosques del límite Santigao del Estero-Chaco , Bosques del Este de Suncho Corral , Planicie aluvial del río Bermejo , Esteros salobres del norte de Santiago del Estero , Área del límite entre Tucumán y Santiago del Estero, Delta del Rio Dulce , Los Bajos submeridionales , Región del Iberá y Ñeembucú , Salinas Grandes, de Ambargasta y otras)* and four to the Pampas (VPA) priority areas (*Cuenca de Laguna la Picasa, Pajonales de paja colorada de la pampa deprimida, Cerrilladas- Llanura periserrana del Sistema de Tandilla, Pastizales del Chasico-Villa Iris*).

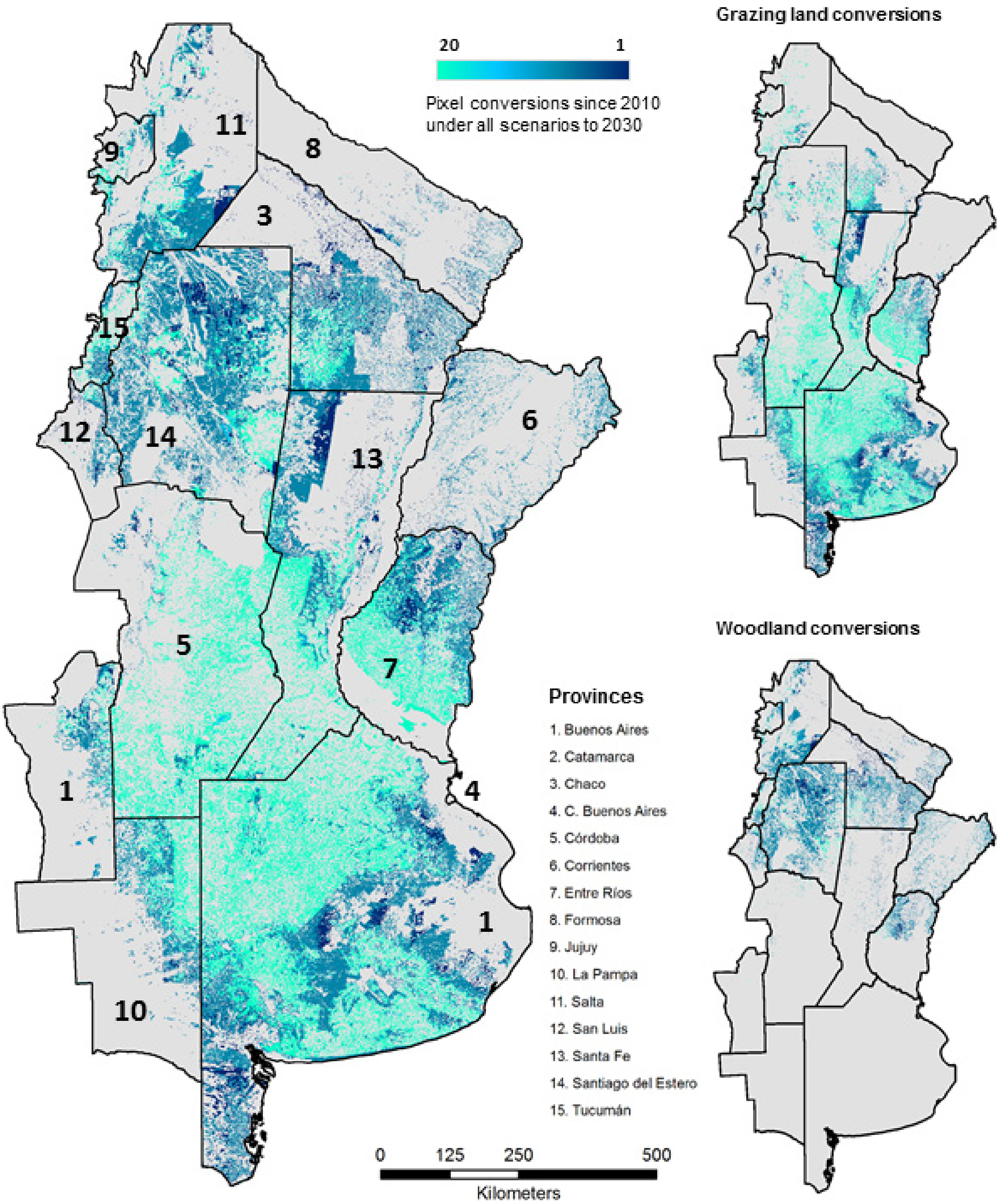

Figure III-4: Frequency of agricultural land-use conversions across all baseline scenarios (BS1, BS2, BS3, BS4) and policies incentives simulated (PI-1, PI-2, PI-3, PI-4).

4 Discussion

Understanding how future agricultural land-use change and how policies may affect these changes is important in light of the environmental trade-offs of agriculture and the increasing future demand for agricultural products (Angelsen, 2010; Schmitz et al., 2014). We here explored potential future agricultural land-use scenarios in northern Argentina under diverging economic policies and identified areas where land-use pressure on conservation priority areas may be high. Our study provides four major insights. First, assuming that land users maximize profits, agricultural land-use change in the Chaco, Espinal and Pampas would shift onto an intensification pathway until 2030, while deforestation would slow down considerably. Second, these patterns remain comparatively unaffected even under fairly drastic policy interventions impacting the profitability of farming. Third, assuming future land use changes at rates of the past increased the impact of our policies, depending on the type of policy intervention, with road improvement to the provincial capitals having the strongest impact. Fourth, our results suggest that the impact of policy interventions varies regionally, and was strongest in regions with a long agricultural history. Thus, our study provides a cautionary view regarding the power of policies targeting profits (e.g., taxes, subsidies) for future agricultural land use in Argentina and highlights the value of scenario-based land-use simulations.

Our baseline scenarios assuming profit maximizing land users (*BS1* and *BS2*) showed little agricultural expansion, but a tendency towards agricultural intensification. From a profit-maximizing perspective this makes sense, as it is much more profitable to converting grazing lands into croplands compared to converting woodlands into croplands, as the latter is much more capital intensive and dependent on environmental characteristics (Dalla-Nora et al., 2014; Henderson et al., 2013; Piquer-Rodríguez et al., in review). Still, these overall low rates were initially surprising to us, given the high past woodland conversion rates, providing further evidence that factors besides those affecting marginal profits (e.g., such as cultural values or the agglomeration of economies) appear to be important in driving land-use change (Garrett et al., 2013; Gasparri et al., 2015; Henderson et al., 2013). Furthermore, we found that agricultural intensification occurred in a clustered way, suggesting a progressing frontier outwards from areas of already high agricultural intensity (Figure III-2). This was expected, as neighbourhood relationships are an important factor explaining land conversions in the region (Piquer-Rodríguez et al., in review; Volante et al., 2016), and established centres of agriculture are characterized by an accumulation of capital, better infrastructure, and technology (Krugman, 1991; Porter, 1998). In such regions (e.g., the Pampas region and the *Anta* and *Charata* regions in the Chaco), land-users can react rapidly to new economic opportunities and our simulations show that intensification from cattle ranching to cropping would occur there quickly, whereas more marginal regions far away from facilities and markets, do respond slower to economic incentives.

Our simulations demonstrated that future land conversion trends and patterns were overall surprisingly unaffected by policy interventions – even under fairly strong policy assumptions. This can be justified by three factors that are important in explaining land-use conversions as in our NRM. First, the economic incentive to convert the land compared to maintaining woodland is very high for both agricultural activities (cropland or grazing) and thus small changes in profits may have little influence in land conversions since it is already very profitable to invest. Second, neighboring effects are important in clustering agricultural activities in the region (Piquer-Rodríguez et al., in review; Volante et al., 2016) and thus patterns arising from our simulations cluster around existing uses. Third, our results suggest a range of other factors not included in our NRM influencing land conversions (such as cultural ties to the land, or agglomeration economies) which may impact future conversions in more diverging land-use trajectories (Garrett et al., 2013; Gasparri et al., 2015; Gasparri and le Polain de Waroux, 2015; Henderson et al., 2013). Our study thus overall provides further evidence for the often limited power that economic policy instruments, such as taxes, subsidies and payment for ecosystem services programmes, may have in influencing strategic land use changes, similar to what was found for the United States (Lawler et al., 2014; Radeloff et al., 2012) or Europe (Stürck et al., 2015).

The impact of our policy options was larger when extrapolating historical land-use change rates (*BS3* and *BS4*). Exploring the effect of policies that would decrease agricultural profits (PI-1) resulted in less cropland expansion into marginal areas, a pattern that can be expected as land rents would be lowered particularly in such areas (Figure III-2), or in areas with longer agricultural history where investments into intensification are less likely under lower profits. At the same time, however, other areas experienced higher woodland to grazing land conversions. One rationale explaining this is the fact that agricultural actors may use their decreasing profits to secure land rights and only establish grazing lands for cattle (Gasparri and le Polain de Waroux, 2015), which is overall less costly and more resilient to climate changes than expanding croplands (Houspanossian et al., 2016; Murray et al., 2016). To the contrary, increasing profits from agriculture (PI-2), for example through lowering taxes, suggests agricultural intensification would increase, since agricultural intensification in our study region is very responsive to marginal profit changes (Piquer-Rodríguez et al., in review). Lastly, our road development scenarios (PI-3 and PI-4) highlight the importance of provincial capitals as regional hubs for agriculture, similar to other regions in the world (Ferretti-Gallon and Busch, 2014; Leblois et al., 2017)

. This suggests that future infrastructure developments plans of Argentina (e.g. *Plan Belgrano*) should be implemented very carefully, as they may affect agricultural expansion patterns strongly (Camara Argentina de la Construccion et al., 2000).

Despite the overall limited impact of our economic policies, important regional variation emerged. For example, policies increasing profits may have the potential to spare forests in marginal regions, as intensification in core agricultural areas already under intensive production would be favored (assuming full flexibility of land users). Conversely, policies lowering profits, could result in an expansion of grazing land into more marginal areas, consistent with the theory that land users would seek to secure land for future agricultural development in such periods (Gasparri and le Polain de Waroux, 2015). Accounting for technological innovation (i.e., yield increases) also had some marked regional effects, potentially lowering the loss of woodland in marginal regions somewhat. Yet, interestingly, yield increases could act as an incentive to further expand agriculture (e.g., south of Salta or north of Santiago) as suggested for other regions (Garrett et al., 2013; Rudel et al., 2009). Such a spatial reorganization is common in places where agriculture industrializes at large scales (Byerlee et al., 2014; Kuemmerle et al., 2016).

An important regional finding from our simulation was also that many areas are very likely to experience future agricultural land-use change, regardless of the scenarios and policy options investigated (Figure III-4). This was the case, for example, for Buenos Aires, the north of Santiago del Estero, southern Salta, southern Cordoba, Entre Rios and south-western Chaco. Given the many areas of conservation concern that would be affected by these land-use changes (Figure SI III-3), careful, proactive conservation planning is needed. Conversely, other areas will experience least land-use change under all policy options, such as San Luis, Catamarca or Formosa, potentially highlighting areas that were not responsive to factors influencing conversions included in our model.

Our simulation approach is not without limitations. First, our NRM assumes actor that maximize economic benefit, although there may be other factors driving land-use conversions, such as social and cultural aspects (e.g., attachment to land), access to capital, land speculation, securing land rights or knowledge diffusion (Garrett et al., 2013; Gasparri et al., 2015; Gasparri and le Polain de Waroux, 2015; Henderson et al., 2013). Second, the NRM is a partial equilibrium model. That is, changes in profit to land owners only influence decisions at the margin, but do not feedback on the entire economy. For example, in the scenarios where profit increases the conversion likelihood changes only through changes in land rent. In reality, if wealth accumulates this alone may influence land user's decisions by changing landowner access to capital. Such dynamics are not included in our model. Third, the model used here was parametrized for the entire region and, as such, did not include the influence of provincial natural resources management policies. For example, we did not project any further grazing land expansion in eastern Salta, that is environmentally suitable for ranching and provincial zonation permits sustainable grazing activities, since grazing land expansion was located in other regions of better suitability. Also unexpected was the relative stability of the Formosa province, where the agricultural frontier has only recently been activated following the paving of road 81 (in 2008, the last two years in our model parametrization period). Fourth, we only assessed quite general policy interventions, and existing policies (e.g. taxes, production caps, etc.) may be changed by the new Argentine government. However, given that our results were relatively unaffected by small policy changes, this should not affect any of our conclusions.

5 Conclusion

Understanding future and use and how policies may alter it is important to avoiding the unwanted outcomes of agricultural expansion and intensification. Using a spatial net returns model, we here show how land-use change in northern Argentina is driven by factors affecting profit, and provide some evidence for the usefulness of economic policy interventions and infrastructure development to alter future land-use change trajectories and patterns. However, our study also showed that the impact of these policies was overall quite limited, and generally stronger in prime agricultural regions. Moreover, our work highlighted that the factors other than those affecting profits at the margin are important driving land-use change in northern Argentina (e.g., cultural values, agglomeration factors, actors securing land rights). Together, this suggests that the power of policies targeting profits (e.g., taxes, subsidies, PES) may alter future land-use trends less than often assumed. Our study also highlighted that many areas of conservation priority will receive high conversion pressure in the future, highlighting the need for ramping up conservation actions. Zoning, as already in place, and the expansion of Argentina's protected area network are likely more powerful tools in avoiding the loss of areas of conservation concern than economic policy incentives. More generally, our study shows how evaluating potential future impacts of economic policies on land-use change rates and patterns may help to inform spatial and conservation planning to steer development pathways towards desired directions.

Acknowledgements

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Supplementary Information

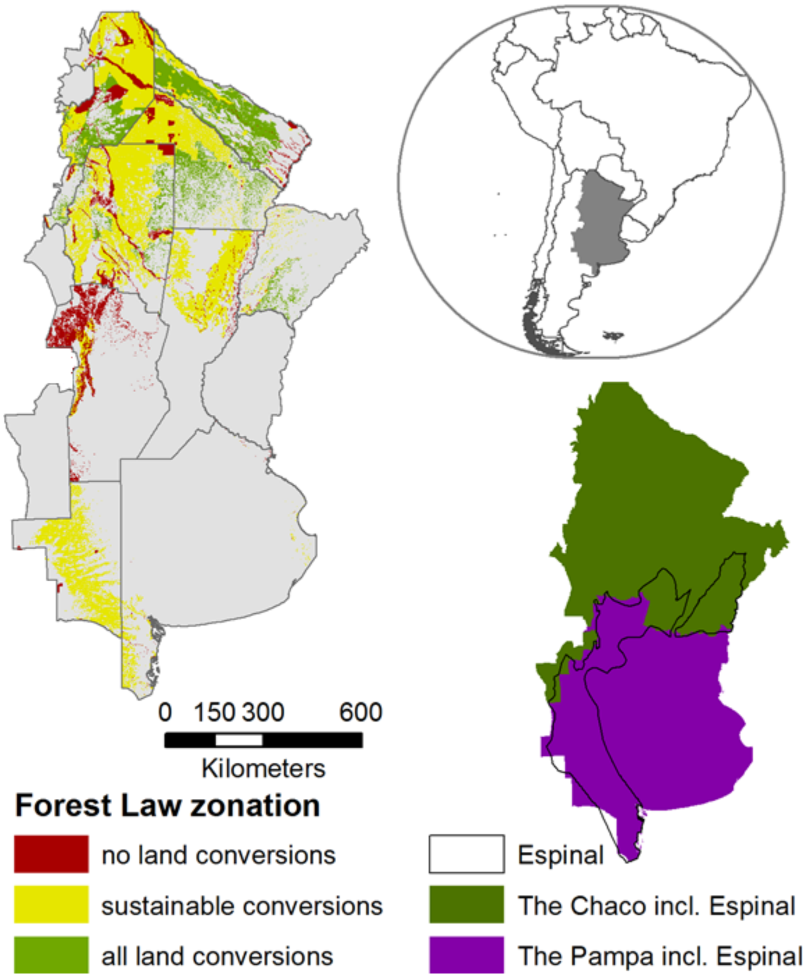

Figure SI III-1::Location of study area in Argentina with the Chaco and Pampas ecoregions that include the Espinal and the zonation of the Forest Law in Argentina, source: UMSEF (2012)

Text SI III-1: Net Returns Model

We modeled three agricultural land-use/cover changes for our study region using two net returns models. First, we used a logit model to assess the conversion of grazing land to cropland in the Chaco and Pampas. Second, we used a multinomial logit model to assess the conversion of woodland to cropland and woodland to grazing land in the Chaco. Using this probabilistic framework, we estimated the likelihood of a parcel of land (i.e., one pixel of 1x1 km^2) converting from woodland to either grazing land or cropland, or converting from grazing land to cropland based on equation III-1 (below) and using our dependent and independent variables (Table SI III-1). For more details see (Piquer-Rodríguez et al., in review).

$$Y^* = B_0 + B_1\, province + B_2 * 1cropneighbor2000 + B_3 * cropneighbor2000 + B_4 * cropprofit + B_{5-8} * soil + B_9 * chaco + B_{10} * grazeprofit + B_{11} * slope + B_{12} * distcapital + B_{13} * \%crop2000 + B_{14} * aridity + B_{15-22} * Interactions + e_i \quad \text{(III-1)}$$

Where Y^* is the latent variable and the error term is distributed with a standard logistic distribution, e~Logistic (0, 1). *Cropprofit* and *grazeprofit* represented average profits for each use at the district level and were calculated as in equation III-2 (below).

$$\text{Average profit to cropland} = \Sigma_{all\ crops}(\%crop_i) * ((yieldcrop_i * pricecrop_i) - costcrop_i) \quad \text{(III-2)}$$

Where *%crop is* the percentage of agricultural land in a district in a given crop*, pricecrop is the* national average price for main crop types, *yieldcrop is* district data on crop yields and *costcrop* the cost to produce each crop summarized at the ecoregion level (Table SI III-1). To calculate average profits to grazing land per district, we used district-level live meat yield data (*yieldmeat*), multiplied this by the national internal producer price (pricemeat), and subtracted the direct costs of production at the ecoregion scale (*costmeat*) (Table SI III-1).

Table SI III-1: Description of variables used to parameterize the net returns model.

Variables		Description	Units	Original scale	Sources
Land-use/cover Conversions	Grazing land to Cropland	Conversions from grazing land to cropland	0-1	$1km^2$	Volante et al. 2015, own data
	Woodland to Grazing land	Conversions from woodland to grazing land	0-1	$1km^2$	Hansen et al. 2013, own data
	Woodland to Cropland	Conversions from woodland to cropland	0-1	$1km^2$	Hansen et al. 2013, Volante et al. 2015
Environmental	*Aridity*	PP/PEVT in 2010	-	$1km^2$	INTA weather stations
	Soil	FAO's index of soil agricultural productivity	0-4	$1km^2$	Atlas de suelos, INTA
	Slope	Degrees of slope	degree	$1km^2$	www.landcover.org (SRTM)
Economic	*pricecrop*	Producer prices at the first point of sale	USD /t (current $)	Country	FAO stats
	pricemeat	Live meat price	USD/t (current $)	Country	FAO stats
	Yieldcrop, yield meat	Crop yields meat produced	t/ha	Department	Databases Integrated System of Agricultural Information (SIIA in Spanish) and Stock cattle INTA2010
	Costcrop, costmeat	Direct costs for crop and meat production	USD/ha (current $)	Ecoregion	INTA, Margenes Agropecuarios, MAyG
	Distcapitals	Cost distance to provincial capitals using roads in 2010	USD (current $)	$1km^2$	IGN-SIG250
Structural	Protected Areas	Network of Protected Areas	0-1	Country	World Database on Protected Areas, www.wdpa.org
	Provinces	Control variable (dummy)	character	Province	Database of Global Administrative Areas
	Ecoregion	Control variable (dummy)	1,2	Ecoregion	WWF
	%cropland2000	Crop area per department in 2000	ha	Department	self-generated
	1cropneighbor2000	None or >=1 crop neighbors in 2000	0,1	$1km^2$	self-generated
	cropneighbor2000	Number of cropland neighbors in 2000	0-8	$1km^2$	self-generated

Text SI III-2: Baseline scenarios

We simulated four baseline scenarios. First, we simulated future agricultural land-use changes assuming that land users seek to maximize land utility, as assumed in our NRM (baseline scenario 1: maximizing land utility –*BS1*) (Table III-1). This scenario assumed that where land was used primarily as an input to production, utility could be described well by economic net returns (i.e., profit or loss). For our second baseline scenario, we altered BS1, which assumed constant agricultural productivity, to add increasing crop yields (baseline scenario 2: maximizing land utility with increasing yields –*BS2*). To do so, we assumed agricultural production targets of the Strategic Food and Agricultural Plan (MAGyP, 2011) to be met by improving crop yields, for example via technological innovation (e.g., new crop varieties such as drought tolerant soybean) or improved management practices (e.g., more efficient water use). Specifically, sunflower yields would increase until 2020 up to 44%, corn 24%, soya 14%, wheat 21%, cotton 37% and sorghum 14%. This would translate into higher total crop production.

We added two more baseline scenario to account for the decision making, and thus agricultural expansion and intensification, possibly being influenced by more factors than the economic, profit-oriented causes entailed in our NRM (Dalla-Nora et al., 2014; Dent et al., 1995; Henderson et al., 2013). Such factors may in the case of Northern Argentina include, for example, land speculation, access to land, capital availability, ties to the land by indigenous communities, or corruption – all processes that cannot easily be captured in an economic modelling framework. Thus, we assumed agricultural expansion and intensification would continue at the rates observed during 2000-2010, when factors other than profit-oriented were possibly important (Arima, 2016; Piquer-Rodríguez et al., in review). In our third baseline scenario, we simulated future land-use conversions assuming historical rates of land-use conversions, as in 2000-2010, and constant yields (baseline scenario 3: historic rates 2000-2010 – *BS3*), whereas our fourth baseline scenario altered scenario BS3 and assumed the same yield increases as in scenario BS2 (baseline scenario 4: historic rates 2000-2010 with increasing yields – *BS4*).

Text SI III-3: Economic policy interventions

Our first policy intervention (*PI-1*) assumed a decrease in the profits for cropping and ranching of 50%. Economic policies leading to such a profit decline could include increasing or imposing new taxes on production exports (*retenciones)* or raising export caps (*ROE*). This has occurred in the past when soy export taxes (*retenciones*) showed ten-fold increases between 2002 and 2008, or when meat export taxes tripled in 2005 (Fernández, 2014; Passaniti, 2011).

Our second policy intervention (*PI-2*) assumed the opposite, that is, a 50% increase in the profits from cropping and ranching activities. Such increases could result; for example, from removing export caps, such as is currently the case for wheat and beef exports (Res. MAGP N° 4/2017, published 03.02.2017, Disp. MAGP N° 6/2015). Likewise, production or export taxes could be lowered, as was the case following the recent government change in Argentina when export taxes for all grain commodities and meat were completely removed, and export taxes for soybean were lowered by 2 percentage points (Decree 133/2015).

A more indirect way for the Argentine government than taxes or subsidies to influence the agricultural sector are investments into infrastructure. Investments in transportation and storage infrastructure, especially road building and paving, play an important role in driving agricultural land-use change, because better infrastructure lowers transportation costs for agricultural commodities substantially, thereby raising the profitability for agriculture in formerly marginal areas (Alves, 2002; Arima, 2016; Gasparri et al., 2015; Pfaff, 1999). In terms of investments in infrastructure, we evaluated two contrasting options. As our third policy option (*PI-3*), we assumed road improvement to happen in the vicinity of provincial capitals and towns located at agricultural expansion frontiers, with the goal to connect these capitals and towns among each other, to boost regional markets. We assumed that capitals would act as regional markets hubs centralizing commercial activities and coordinating transportation to export hubs (e.g., harbors). To do so, we chose the roads with the least travel time among them, and assumed all roads within a distance of 100 km from capitals and towns to become paved. This scenario resulted in a total of additional 2,000 km paved roads (Figure SI III-2). Our fourth and last policy option (*PI-4*), assumed that road paving would take place in the vicinity of all towns larger 50,000 inhabitants (including capitals). In this scenario, we assumed that investments into infrastructure would focus on connecting these towns with the main export harbors of Buenos Aires and Santa Fe. This resulted in a total of 8,700 km of additional paved roads.

We assumed all road pavements in our third and fourth policy option to happen between 2010 and 2020 (Figure SI III-2). Both of these scenarios are plausible, as the Federal Council Network of Argentina (*Consejo Vial Federal*) planned to pave 6,310 km of provincial roads until 2024 (Consejo Vial Federal Argentina, 2014). Likewise, 2,800 km of national roads in Argentina are planned to be paved until 2025 (Bortolín, 2015).

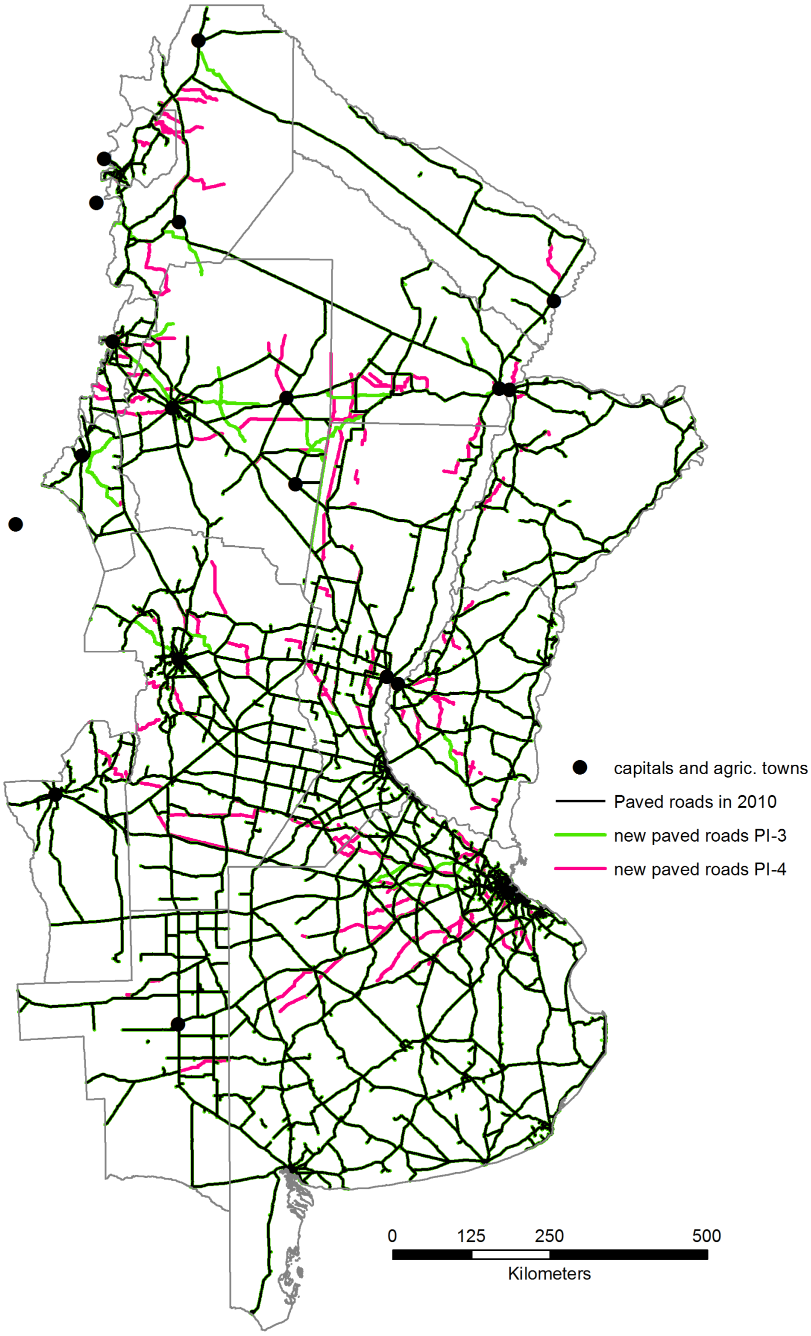

Figure SI III-2: New roads paved under policy intervention 3 (PI-3) and 4 (PI-4). New roads paved under PI-4 also include those in PI-3.

Text SI III-4: Regions of conservation concern

The following priority areas for conservation (Bilenca and Miñarro, 2002; TNC, 2005) were identified as of conservation concern by our analysis because they were located within regions of high likelihood of experiencing agricultural conversions in Argentina in the future (Figure SI III-3).

1. Transición Chaco-Yungas (TNC): Jujuy, Salta, Tucuman, 3,000km^2 (partially affected)
2. Bañados del Quirquincho (TNC): Salta, 2,700km^2 (greatly affected)
3. Zona del impenetrable (TNC): east of Salta, 280km^2 (greatly affected)
4. Derrames de los ríos Hornones y Ureña (TNC): west of Santiago del Estero, 1,700km^2 (greatly affected)
5. Bañados del río Salado y Bañados de Figueroa (TNC): west of Santiago del Estero, 420km^2 (greatly affected)
6. Bosques del límite Santigao del Estero-Chaco (TNC): east of Santiago del Estero, west of Chaco 8,000km^2 (partially affected)
7. Bosques del Este de Suncho Corral (TNC): Santiago del Estero 1,000km^2 (greatly affected)
8. Planicie aluvial del río Bermejo (TNC): east of Chaco 1,000km^2 (partially affected)
9. Esteros salobres del norte de Santiago del Estero (TNC): north Santiago del Estero, 1,080km^2 (partially affected)
10. Área del límite entre Tucumán y Santiago del Estero, al sur embalse río Hondo (TNC): west of Santigo del Estero 1,600km^2 (greatly affected)
11. Delta del Rio Dulce (TNC): east Santiago del Estero, 840km^2 (partially affected)
12. Los Bajos submeridionales (TNC): Santa Fe, 34,600km^2 (partially affected)
13. Región del Iberá y Ñeembucú (TNC): Corrientes 30,500 km^2 (slightly affected)
14. Salinas Grandes, de Ambargasta y otras (TNC): Cordoba and Santiago del Estero 20,000 km^2 (slightly affected)
15. Laguna Mar Chiquita (TNC): Cordoba and Santiago del Estero 17,500 km^2 (slightly affected)
16. Cuenca de Laguna la Picasa (VPA): Cordoba-Santa Fe-Buenos Aires, 5,500km^2 (greatly affected)
17. Pajonales de paja colorada de la pampa deprimida (VPA): Buenos Aires, 22,700km^2 (greatly affected)

18. Cerrilladas- Llanura periserrana del Sistema de Tandilla (VPA): Buenos Aires, 13,800km^2 (greatly affected)
19. Pastizales del Chasico-Villa Iris (VPA): south Buenos Aires, 4,000km^2 (greatly affected).

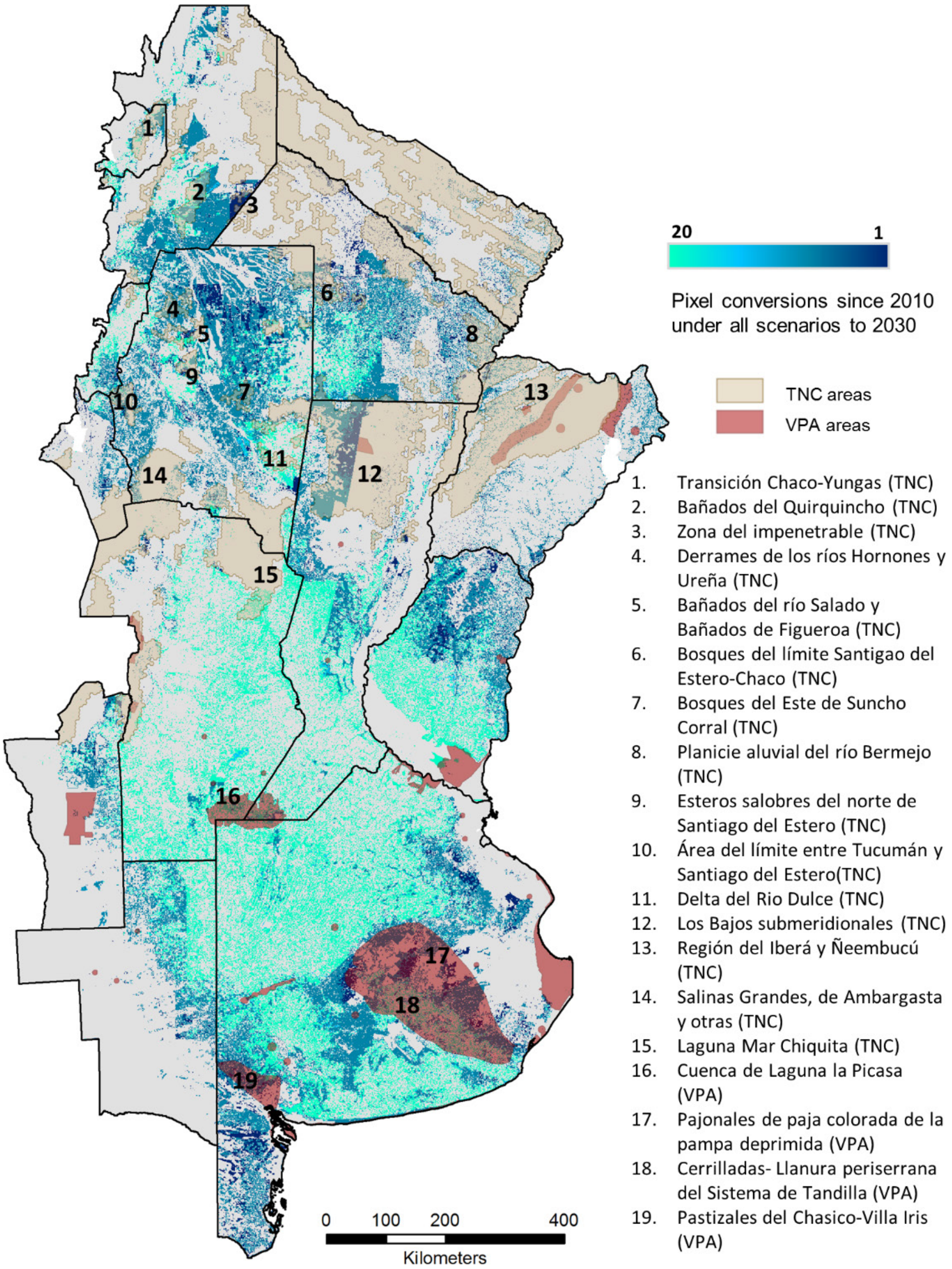

Figure SI III-3: Regions of conservation concern with numbers from 1-19, that lay within or nearby regions with high likelihood of experiencing agricultural conversions in Argentina in the future (in magenta and dark blue). TNC: The Nature Conservancy (TNC, 2005), VPA: Valuable Pasture Areas (Bilenca and Miñarro, 2002).

Chapter IV:
Effects of past and future land conversions on forest connectivity in the Argentine Chaco

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Abstract

Land-use change is the main driver of habitat loss and fragmentation worldwide. The rate of dry forest loss in the South American Chaco is among the highest in the world, mainly due to the expansion of soybean production and cattle ranching. Argentina recently implemented a national zoning plan (i.e., the Forest Law) to reduce further forest loss. However, it is unclear how the effects of past deforestation and the implementation of the Forest Law will affect forest connectivity in the Chaco. Our main goal was to evaluate the potential effect of the Forest Law on forest fragmentation and connectivity in the Argentine Chaco. We studied changes in the extent, fragmentation, and connectivity of forests between 1977 and 2010, by combining agricultural expansion and forest cover maps, and for the future in a scenario analysis. Past agricultural expansion translated into an overall loss of 22.5 % of the Argentine Chaco's forests, with deforestation rates in 2000–2010 up to three times higher than in the 1980s. Forest fragmentation and connectivity loss were highest in 1977–1992, when road construction fragmented large forest patches. Our future scenario analysis showed that if the Forest Law will be implemented as planned, forest area and connectivity in the region will decline drastically. Land-use planning designed to protect stepping stones could substantially mitigate connectivity loss due to deforestation, with the co-benefit of preserving the greatest amount of biodiversity priority areas across all evaluated scenarios. Including scenario analyses that assess forest fragmentation and connectivity at the ecoregion scale is thus important in upcoming revisions of the Argentine Forest Law, and, more generally, in debates about sustainable resource use.

1 Introduction

Land-use change is the main driver of habitat loss and fragmentation (CBD, 2010; Sala et al., 2000), thereby threatening many species (Barnosky, 2008; Ehrlich and Pringle, 2008; Vignieri, 2014). While some species can persist in fragmented landscapes, or even benefit from fragmentation, many species become more vulnerable because their populations are smaller (Cagnolo et al., 2006), they are more prone to overexploitation (Bennett and Saunders, 2010; Michalski and Peres, 2005) and edge effects (Gascon et al., 2000; Lopez de Casenave et al., 1995), and their capacity to adapt to environmental change is lower (Brook et al., 2008; Travis, 2003). Preserving or restoring connectivity is therefore increasingly recognized as a key goal for land-use and conservation planning (Vos et al., 2008).

Understanding how land-use change affects connectivity at the ecoregional scale is particularly important, because many species are endemic at this scale, and ensuring populations' persistence is therefore critical. Yet conservation planners face substantial challenges when managing for ecoregional connectivity. First, many ecoregions are very large or extend across jurisdictional boundaries, and decentralized land-use and conservation planning (e.g., province- or national-scale planning for ecoregions extending into several countries) can have unintended results at the aggregate, ecoregional scale. Second, understanding whether land-use or conservation policies implemented to maintain or improve landscape connectivity will continue to work out as intended in the future is challenging, given uncertain future land-use patterns (Faleiro et al., 2013; Piquer-Rodríguez et al., 2015). Exploring how land-use change may affect landscape fragmentation and connectivity under alternative future scenarios can be a powerful tool to inform conservation planning, yet such assessments are scarce (Ernst, 2014a; Piquer-Rodríguez et al., 2012; Rubio et al., 2012).

South America has recently experienced widespread forest loss (Grau and Aide, 2008), especially in the Cerrado and Gran Chaco ecoregions (Olson et al., 2001) which had among the highest deforestation rates worldwide between 2000 and 2010 (Aide et al., 2013; Hansen et al., 2013). Deforestation during this period was predominantly driven by the rapid expansion of agri-business farming, principally soybean and intensified cattle ranching (Aide et al., 2013; Gasparri and Grau, 2009; Klink and Machado, 2005; Zak et al., 2004). Argentina increased soybean production by 78 % between 2000 and 2010 (Nassar and Barcellos Antoniazzi, 2011). This resulted in considerable soybean and cattle ranching expansion into natural ecosystems (Adamoli et al., 2011; Aizen et al., 2009; Gasparri et al., 2013; Mastrangelo and Gavin, 2012). Pressure on remaining forests is likely to remain high, due to increasing global soybean demand (Diogo et al., 2014; Lambin and Meyfroidt, 2011; Reenberg and Fenger, 2011).

Deforestation threatens the rich biodiversity of the Argentine Chaco, which includes 145 mammal species (12 endemic), 409 birds (7), 54 reptiles (17), 34 amphibians (8), and more than 80 plant genera (3,400 species, of which 400 are endemic) (Bucher and Huszar, 1999; Giménez et al., 2011). The Chaco is also a globally significant carbon pool (Gasparri et al., 2008). Yet, only a few studies have so far assessed forest loss and fragmentation in the Argentine Chaco, generally focusing on small regions (Boletta et al., 2006; Gasparri et al., 2010; Grau and Aide, 2008; Grau et al., 2005b; Torrella et al., 2013; Volante et al., 2012; Zak et al., 2008) or relatively short time periods (Aide et al., 2013; Caldas et al., 2013; Clark et al., 2010; Hansen et al., 2013; Portillo-Quintero and Sánchez-Azofeifa, 2010).

Concern about protecting remaining native forests in the Argentine Chaco led to a national Forest Law, passed in 2007 (*Ley de Presupuestos Mínimos de Protección Ambiental de los Bosques Nativos* 26.331). The Forest Law zones forest areas into three different classes of land-use restrictions: class 1 ('red zones') allows no commercial use; class 2 ('yellow zones') allows only sustainable uses; and class 3 ('green zones') allows most land uses, including deforestation for agricultural expansion [for more details see Seghezzo et al. (2011) and García Collazo et al. (2013)]. The Forest Law was planned in a decentralized way (i.e., each province developed their own zonation and implementation framework) and large spatial discrepancies exist when comparing provincial plans (see Figure IV-1).

The potential effect of the Forest Law on forest connectivity at the ecoregion scale has neither been considered nor assessed. A better understanding of how current implementation of the Forest Law may affect the Chaco's forests is therefore important for informing regional planning. Our main goals were to evaluate past and potential future changes in the extent, fragmentation, and connectivity of forests in the Argentine Chaco. Specifically, our research questions were:

1. What was the influence of past deforestation on forest extent, fragmentation, and connectivity in the Argentine Chaco?

2. How will forest extent, fragmentation, and connectivity of the Argentine Chaco develop if the deforestation allowed under the Forest Law takes place?

3. What would be the potential effect of ecoregional conservation strategies to mitigate further loss of forest connectivity?

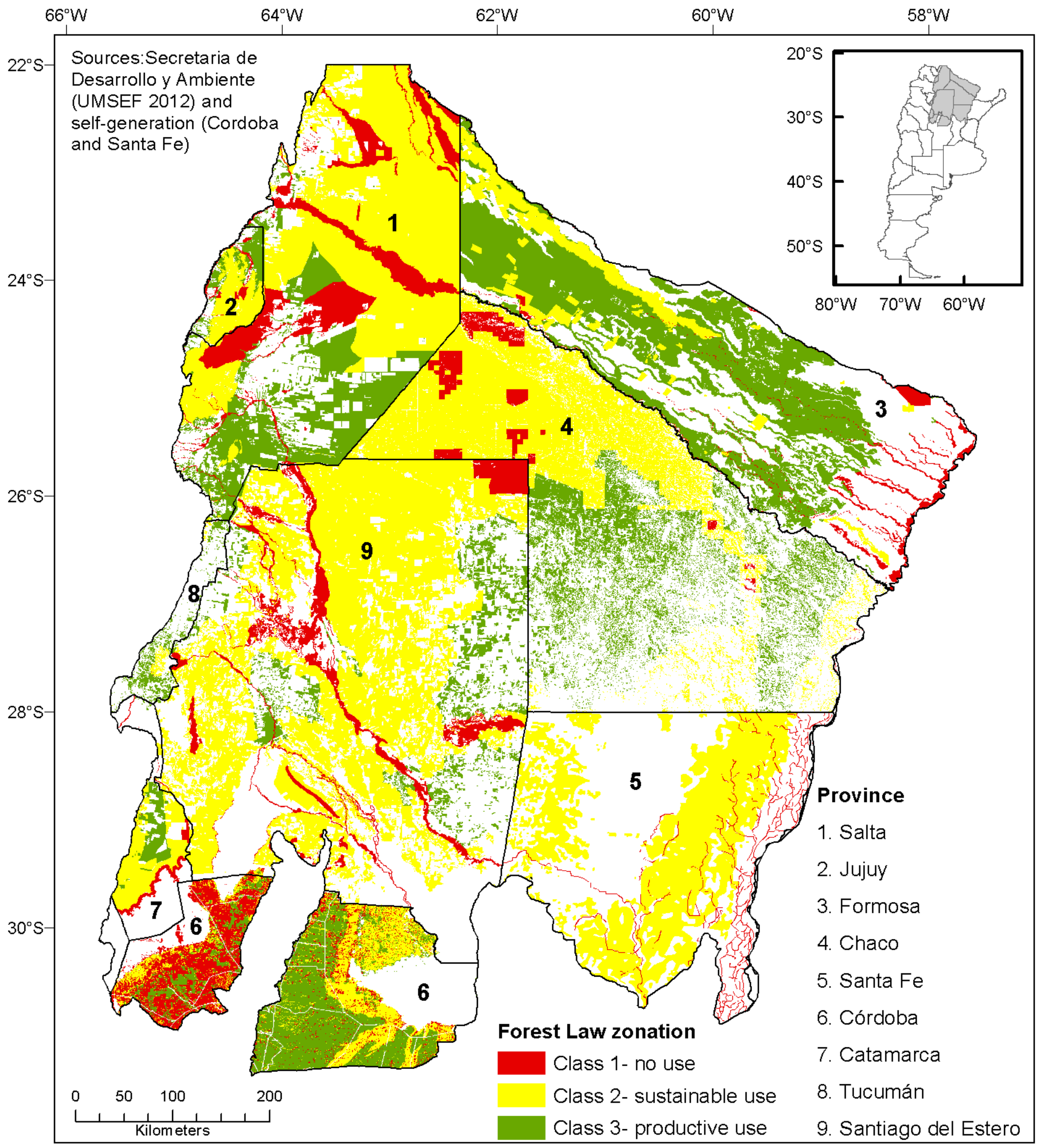

Figure IV-1: Location of study area within the Dry and Wet Chaco and the current zonation of the Argentine Forest Law [we excluded the small mountainous Chaco, (Brown and Pacheco, 2006)]. Class 3 (green) allows commercial uses of the forest, including its conversion to agriculture. Class 2 (yellow) allows sustainable forest uses, and class 1 (red) fully protects forests except for the province of Cordoba, where exceptions are possible. White background color stands for non-forested areas and forests that were not zoned (García Collazo et al., 2013).

2 Study Area

The Gran Chaco is a large, dry forest region covering about 1,080,000 km2 in Argentina (60 % of the Gran Chaco), Bolivia (11 %), Paraguay (28 %), and Brazil (1 %) (Olson et al., 2001). The climate is semi-arid and highly seasonal, with a distinct dry season in autumn and winter (May–September), and a warm, wet season in spring and summer (November–April). Mean annual temperature is ~22 °C, with an average monthly maximum of 28 °C (Minetti, 1999). Annual precipitation ranges from 1,200 mm in the east (wet Chaco) to 450 mm in the west (dry Chaco). Elevation varies marginally except for the west and southwest of the study area where more hilly terrain prevails. Natural vegetation in the Chaco consists of closed forest, open woodlands, shrublands, and palm savannas. Forests are the most characteristic vegetation formation and are typically dominated by species of the genera Schinopsis and Aspidosperma ("quebrachos") (Prado, 1993). Here, we focus on the entire Argentine Dry and Wet Chaco, except for the small mountainous Chaco areas (Brown and Pacheco, 2006) (Figure IV-1).

Traditionally, the dominant land use in the Chaco has been subsistence agriculture and extensive cattle ranching in so-called puesto (small homestead) systems (Grau et al., 2008). Recently, native forest and puesto systems have increasingly been replaced by agri-business farming, mainly for growing cash crops such as soybeans, sugarcane, maize, and cotton (Adamoli et al., 2011; Grau et al., 2005a; Grau et al., 2008; Volante et al., 2006). Likewise, low-intensity grazing and natural forests are increasingly being replaced by intensive silvopastoral systems that clear the understory and plant exotic, more productive grasses (e.g., Cenchrus ciliaris or Panicum maximum) (Macchi et al., 2013).

3 Materials and Methods

3.1 Datasets used

We used base maps of forest cover generated from satellite-based maps of agricultural (cropland and pasture) extent and expansion (available from Adamoli et al. (2011)) and a natural vegetation map (SAyDS, 2007). Agricultural extent and expansion were digitized from Landsat imagery for the base years 1977, 1992, 2002 and 2010 with a minimum mapping unit (MMU) of 0.02km^2 (Adamoli et al., 2011). Because these maps did not provide information on forest extent, nor on the type of vegetation replaced by agricultural expansion, we used the Argentine forest inventory map together with ancillary data (roads and rivers) to derive forest maps for the base years. The National Forest Inventory map (SAyDS, 2007) is a base map of natural vegetation at a scale of 1:100,000 produced from 1997 Landsat image interpretation using a MMU of $0.1km^2$, distinguishing 20 vegetation classes, including forests, shrubs, and grasslands. We reclassified this map into a forest (i.e., all woody vegetation) and non-forest map (Table IV-1). We included shrublands and open woodlands in our woody-vegetation definition because they constitute important types of natural vegetation in the Chaco as a whole (Tálamo et al., 2012).

Table IV-1: Woody vegetation types of the Argentine Forest Inventory reclassified as forest habitats for our study(closed and open woodland or shrubland) and the percentage of each vegetation type under each class of the forest law zonation

		Forest Law categories (area %)		
Forest inventory description	Forest habitats	1-Red	2-Yellow	3-Green
Tall and dense Quebrachal	Closed woodland	1.0	1.2	0.7
Predominance of Quebracho colorado and Blanco	Closed woodland	15.2	27.1	27.2
Quebrachal with discontinuous cover	Open woodland	17.6	20.1	8.7
Quebrachal and other valuable species	Closed woodland	1.2	4.2	8.7
Open quebrachal and other species	Open woodland	0.1	1.4	0.3
Riparian clustered forest	Open woodland	1.8	0.5	0.1
Islet forest	Open woodland	0.1	0.2	0.5
Dominance of Vinal, Mistol or Itin	Open woodland	3.3	4.5	11.1
Bushes with Horco-Quebracho	Shrubland	1.1	0.1	0.2
Forest cover of 50-74%	Closed woodland	0.0	0.0	0.0
Forest cover of 25-49%	Open woodland	0.6	0.6	0.5
Forest cover of 10-24% (minus agriculture)	Open woodland	1.1	1.3	4.7
Low bushes and herbaceous plants	Shrubland	3.6	6.4	5.2
Bushes with or without few trees	Shrubland	5.8	7.2	3.8
Plains with herbaceous plants and bushes	Shrubland	0.0	0.0	1.3
Salty complexes				
Banyados (halophytic vegetation)				
Natural pastures				
Esteros (partly flooded)				
Hidrophylic herbaceous vegetation				

To reconstruct forest/non-forest maps for 1977 and 1992, we reclassified all agricultural expansion in the periods 1977–1992 and 1992–2002 as forest for the respective base year with a MMU of $0.1km^2$. To reconstruct the forest/non-forest maps for 2002 and 2010, we simply erased the agricultural expansion area for each period from the forest inventory map. We also erased 100m around main roads of 1985 and 2005, and along main rivers to account for road margins and river widths that were not captured in the maps (SIG250, http://www.ign.gob.ar/sig#descarga).

As study region boundaries, we used province boundaries from the Database of Global Administrative Areas (www.gadm.org). Because forest patches can extend beyond national boundaries, we extended our study area by a buffer of 20km to avoid distorting effects in the fragmentation and connectivity analyses (e.g., larger forest patches split into several smaller ones, etc.). The 20-km buffer was selected because it roughly represents the diameter of the home range of the two most wide-ranging apex predators in the Chaco, the puma (*Puma concolor*) and the jaguar (*Panthera onca*) (Canevari and Vaccaro, 2007). To extend our forest/non-forest maps into these buffer areas, we used forest maps for Bolivia (SAB, 2001) and Paraguay (The Global Land Cover Facility, 2006). These maps were static, but deforestation in our study period there was negligible (Hansen et al., 2013; Huang et al., 2009; Killeen et al., 2007).

To assess how past and future deforestation affect conservation priority sites, we used the *"Conservation Portfolio of Priority Areas for Biodiversity"* (TNC, 2005) generated at a scale of 1:750,000 for the entire Gran Chaco (Figure SI IV-1). Using multi-criteria analyses and targeted workshops, experts on Chaco wildlife, conservation, and ecology from Argentina, Paraguay, and Bolivia outlined (1) so-called *Areas of Biodiversity Significance* (ABS) for each major taxa (birds, amphibians and reptiles, mammals, and vegetation and plants) based on their regional knowledge and (2) defined conservation planning goals for the Chaco (e.g., minimum area for certain ecosystems). Based on both, experts then established priority areas within the ABS that should be protected to reach the identified conservation goals and that, at the same time, were particularly threatened considering current human pressure (TNC, 2005).

3.2 Mapping past forest change, fragmentation, and connectivity

We calculated forest area change for 1977–1992, 1992–2002, and 2002–2010. To assess forest fragmentation we used Morphological Spatial Pattern Analysis (MSPA, Vogt et al. (2007)). MSPA segments a binary forest/non-forest map into five fragmentation components (Soille and Vogt, 2009): core, bridge (i.e., connections among core areas), islet (i.e., small patches without core forest), edge, and perforation (i.e., edge inside core patches). We calculated forest fragmentation maps for each base year (using an eight-neighbour rule and a one-pixel edge), summarized fragmentation components, and calculated change between the fragmentation maps. Additionally, we calculated the degree of fragmentation in percent, based on entropy theory. Minimum values for this measure are reached when the forest cover is a single compact patch, while maximum values are reached when the number of patches is the maximum possible and patches are dispersed over the entire study region (Vogt, 2014).

We assessed potential forest connectivity by calculating the proximity index (PROX) and the connectance index (CONNECT) (McGarigal et al., 2012) for the forest/non-forest maps. These two metrics have been shown to perform well and complement each other in measuring landscape connectivity and fragmentation (Wang et al., 2014). To parameterize these indices, we used the diameter of the home ranges of intermediate dispersers in the Argentine Chaco (i.e., 2km, Canevari and Vaccaro 2007) as a proxy for the maximum distance between patches that we considered connected. Home ranges are related to dispersal distances (Bowman et al., 2002) and are often used in connectivity analyses to establish movement distances (O'Brien et al., 2006; O'Farrill et al., 2014; Schumaker et al., 2014). Intermediate dispersers benefit most from connecting elements at the landscape scale (Rubio and Saura, 2012) and some examples are the giant armadillo (*Priodontes maximus*), the giant anteater (*Myrmecophaga tridactyla*), and the collared peccary (*Pecari tajacu*), all of which are of conservation concern in the Chaco (Tognelli, 2005). Thus, our connectivity analyses combined elements from both structural (i.e., landscape structure) and functional (i.e., species-focused) connectivity analyses to study potential connectivity [sensu Calabrese and Fagan (2004), Ernst (2014b)].

The PROX index (Equation IV-1) calculates Euclidean distances among patches that are within a specific search radius, in our case forest patches located at a maximum distance of 2 km from the focal patch, while taking into account the size of neighboring patches:

$$\text{PROX}= \sum_{s=1}^{n} \frac{a_{js}}{h_{js}^{2}} \quad \text{(IV-1)}$$

where a_{js} refers to the area of patch js within a specified search radius (2 km in our case) of patch j; j refers to the the jth focal patch and s refers to the sth neighboring patch within the search radius of patch j; h_{js} is the distance between the patch js and the focal patch, based on edge-to-edge distance (McGarigal et al., 2012). PROX thus assigns higher values to patches that are surrounded by many and/or bigger patches, and is preferable over indices that are not area-sensitive (Bender et al., 2003; Fahrig, 2013). Since PROX is calculated at the patch level, we also calculated the mean PROX (i.e., PROX_MN) for each of our maps.

The CONNECT index (Equation (IV-2)) below calculates the proportion of functional joins in the entire landscape given our threshold radius of 2km (McGarigal et al., 2012):

$$\text{CONNECT}=\frac{\sum_{j\neq k}^{n} c_{jk}}{\frac{n_j(n_j-1)}{2}}(100) \quad \text{(IV-2)}$$

where c_{jk} equals the joining between patch j and k (0 = unjoined, 1 = joined), and n equals the number of patches in the landscape. CONNECT equals or is close to zero when the landscape is composed of a single patch or none of the forest patches are connected, and equals 100 when all patches are connected (McGarigal et al., 2012).

As a sensitivity analyses, we performed the connectivity analysis at different MMU scales of 0.05, 0.1 and 0.2 km^2 and our results were not sensitive to the MMU size. Furthermore, we evaluated the sensitivity of our connectivity results to the selected home range diameter.

3.3 Future forest change, fragmentation, and connectivity

To evaluate how the implementation of the Forest Law may influence future forest fragmentation and connectivity in the Chaco, we assessed three sets of scenarios differing in the assumed amount of forest conversion (Table SI IV-1 and Table SI IV-2). Each province specifies a range of forest conversion that can take place in each zone (e.g., 20–60 % in green zones in Formosa), sometimes according to some spatial attributes such as slope (e.g., Salta), plot size (e.g., Chaco) or land-use zonation (e.g., Formosa). Because we were interested in the potential full impact of implementing the Forest Law, we chose the maximum conversion limit specified per province by law as a basis to calculate deforestation amounts for different conservation strategies. Conversion amounts in our scenarios are thus alternative assumptions of how the future may unfold, and should not be interpreted as forecasts of forest loss.

The first base scenario (scenario 1) assumed that all areas assigned for conversion in green zones per province, according to the maximum conversion limits permitted by the Forest Law, will actually be deforested (Table SI IV-1). A second base scenario (scenario 2) assumed that all areas assigned for conversion in green and yellow zones per province, according to the maximum conversion limits permitted by the Forest Law, will be deforested. The case of Cordoba is an exception because conversions can also take place in red zones (Table SI IV-1). This scenario represents a worst-case where conversions take place in green and yellow zones as currently planned (UMSEF, 2012). We refer to this scenario as “planned implementation of the Forest Law”. Finally, a third base scenario (scenario 3) assumed that green and yellow zones will experience deforestation following annual deforestation changes of the last 20 years and up to the maximum conversion limits permitted by the Forest Law (see “Future forest change, fragmentation, and connectivity” section and Table IV-2).

Table IV-2: Deforestation amounts (km^2) implemented under each scenario. Random and systematic (distance to human settlements) stands for the type of allocation scheme of each scenario type. *NP* No protection, *PB* protection of big patches only, *PS* protection of big patches and stepping stones.

	Scenario 1		**Scenario 2**		**Scenario 3**	
	random	*systematic*	*random*	*systematic*	*random*	*systematic*
NP	46,964	46,967	158,879	149,954	46,919	46,924
PB	23,215	23,189	69,658	70,384	41,800	43,947
PS	19,515	19,480	47,182	48,518	27,460	26,350

Each of these three base scenarios were calculated in three versions: (a) without additional conservation measures, (b) assuming the protection of the biggest forest patches (Almeida-Gomes and Rocha, 2014) and (c) assuming additional protection of small patches, potentially acting as stepping stones (Saura et al., 2013). The protection of big patches required that at least 70 % of the biggest patches to be preserved to avoid potentially non-linear and strong increases in population declines and extinction risk (Camargo Martensen et al., 2012; Fahrig, 2003; Swift and Hannon, 2010). To define the biggest patches we evaluated the distribution of patches’ area and selected the 20 biggest patches, each with an area greater than 1,670 km^2. The more restrictive conservation strategy (strategy c) preserved all stepping stones identified as important, as well as a minimum of 60 % of the area of the biggest patches (Table IV-2). To define potential stepping stones, we arbitrarily choose the patches within the top 10 % of PROX index values among patches not considered “big”. We also limited deforestation to be 25 % lower than conversions allowed in the Forest Law and to preserve a minimum of 40 % of the 2010 forest cover to avoid accelerated extinction (Andren, 1994; Fahrig, 2003; Villard and Metzger, 2014), even if the Forest Law would allow for higher conversion limits (Table SI IV-1).

This resulted in three base scenarios (1–3) and three conservation strategies (a: no protection, referred to as *np*; b: protection of big patches only, *pb*; c: protection of big patches and stepping stones, *ps*) and thus a total of nine scenario runs. Scenarios were derived using two alternative allocation procedures: random and systematic. First, we used a tenfold random assignment of deforestation plots until the target deforestation amount per province was reached. To allocate deforestation plots, we generated squared grids with a cell size equal to the median parcel size of the agricultural plots per province, ranging from 0.48 to 5.3 km^2 (INDEC, 2002). Second, the systematic allocation scheme assumed deforestation to only occur at the agricultural frontier. We assumed that plots closer to settlements and infrastructure would have a higher chance to be converted to agriculture because demographic dynamics and access to markets are important drivers of deforestation worldwide (Carr, 2004; Müller et al., 2011). We used the same plots as above and calculated Euclidean distances per plot to the nearest settlements with >100 inhabitants (localities and cities, SIG250). Those forested plots closest to settlements were selected for conversion until the specific conversion amounts per scenario and province were reached (Table IV-2). The choice of these spatial allocation procedures was based on visually inspecting past deforestation patterns, suggesting that both gradual deforestation frontiers as well as leapfrogging processes take place in the study region. We then used the resulting binary forest/non-forest maps to calculate fragmentation and connectivity measures as detailed in the previous section. Furthermore, we derived the average and standard deviation share of fragmentation components across the 10 replicate runs for each scenario (only random allocation scheme).

3.4 Past and future forest loss in priority conservation areas

We compared the past and future forest cover maps with the priority conservation areas identified in the only broad-scale conservation planning exercise carried out for the Chaco so far (TNC, 2005). We summarized forest conversion for the total set of priority areas (i.e., including all priority areas for terrestrial and aquatic ecosystems, birds, amphibians, reptiles, mammals, and vegetation communities) and the terrestrial-only priority set because we specifically studied in-land processes.

4 Results

4.1 Past forest change, fragmentation, and connectivity

The Chaco experienced a dramatic acceleration in deforestation since the start of our study period in 1977 (Figure IV-2). During both 1977–1992 and 1992–2002, about 20,000km^2 were deforested, while almost 40,000 km^2 were deforested in 2002–2010. This equals a rate of roughly 4,750km^2/year (>0.5km^2/h) over the last decade—more than double the rates from the 1990s and more than three times the rates from the 1980s. Our deforestation maps highlighted agricultural expansion frontiers especially in the surroundings of Las Lajitas (Salta Province), Tucuman, Charata (Chaco), Quimili and Bandera (Santiago del Estero) and west of Cordoba. Deforestation tended to occur both along agricultural frontiers and in a more random, leapfrogging way (e.g., in Salta and Formosa Provinces, Figure IV-2).

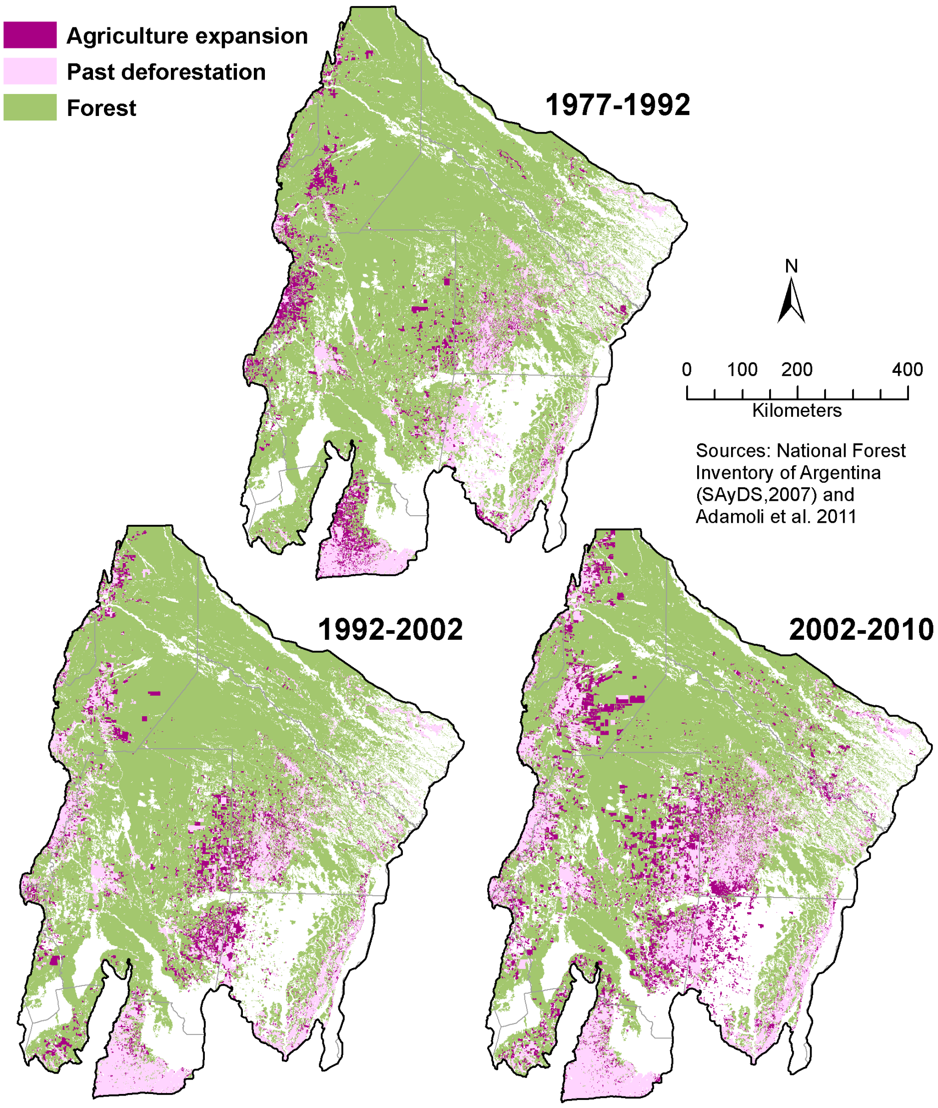

Figure IV-2: Forest maps derived from the National Forest Inventory of Argentina (SAyDS, 2007) and agricultural expansion data (Adamoli et al., 2011). Agriculture area in 1977 is used as the baseline.

Provincial borders of the zonation map of the Forest Law often mark strong inconsistencies in zoning, such as in the case of the border between Salta and Chaco, where forests classified as green and yellow are adjacent, or between Chaco and Santiago del Estero (yellow and red, respectively, Figure IV-1). Yellow zones (sustainable uses) cover by far the greatest area (170,000 km^2), followed by green zones (all uses, 80,000 km^2), and red zones (no use, 36,000 km^2). The *quebrachales* was the dominant vegetation type of the forest inventory protected by the Forest Law (34 % of red zones), but also the most dominantly assigned to sustainable development (48 % of yellow zones) and potential deforestation (37 % of green zones, Table IV-1). About 2,800 km^2 (2.5 %) of forest conversions took place in red zones since the implementation of the forest law (AGN, 2014), mostly in Cordoba (1,900km^2), Salta (300km^2), Santiago del Estero (260km^2), and Santa Fe (235km^2). As a result of deforestation between 1977 and 2010, edge forest increased by 8.2 % and the number of forest patches increased from ~8,000 patches in 1977 to ~15,000 patches in 2010. Bridge forest decreased strongly throughout the period studied, especially in 1977–1992, when many bridges became isolated or were deforested (Figure IV-3). The degree of fragmentation increased accordingly from 1977 to 2010 (Figure SI IV-2).

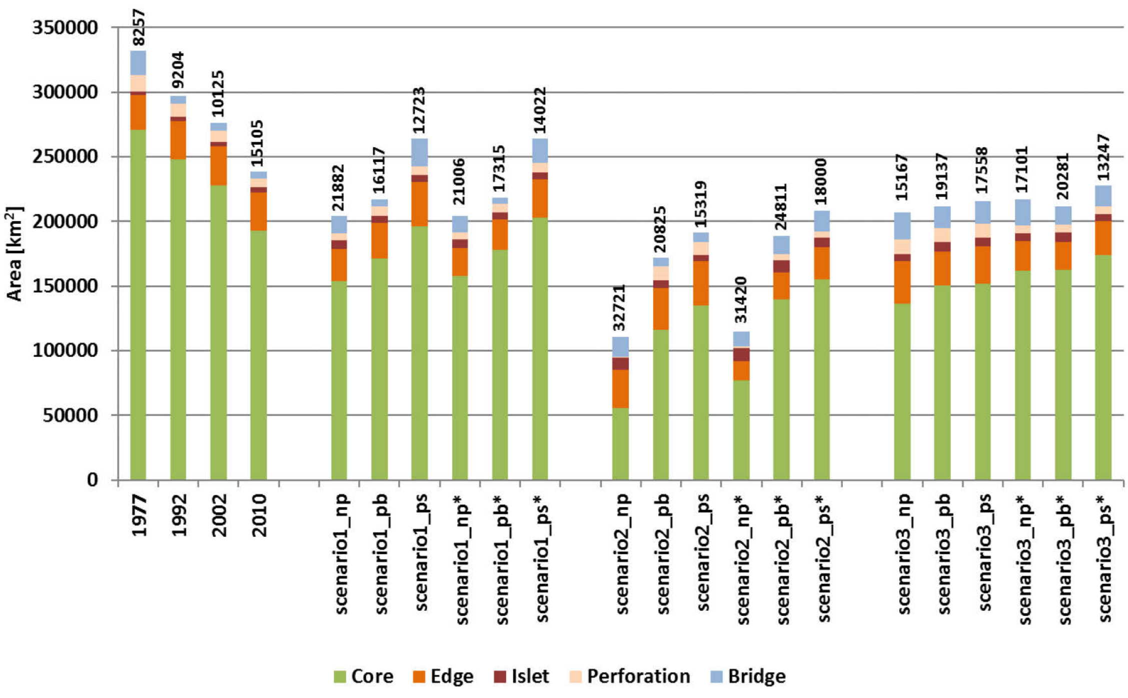

Figure IV-3: Extent of forest in different fragmentation classes for the past and for the future scenarios. *Numbers* on top of the columns are the number of forest patches in the landscape. np= no protection, pb= protection of big patches only, ps = protection of big patches and stepping stones. Scenario 1 allows conversions in *green* zones, scenario 2 allows conversions in *green* and *yellow* zones and scenario 3 allows conversions in *green* and *yellow* zones following historic deforestation amounts. Scenarios marked (*asterisk*) used a systematic allocation scheme for deforestation (distance to human settlements).

Landscape connectivity (CONNECT index) decreased by 27 % from 1977 to 2010 (Figure II-4). Connectivity at the patch level, as measured by the average PROX, decreased by 95 % from 1977 to 2010 (Figure IV-4). The strongest decline in connectivity occurred in 1977–1992 (Figure IV-4). This is in accordance with the high loss of bridges and an increase in edge forest for the same time period (Figure IV-3).

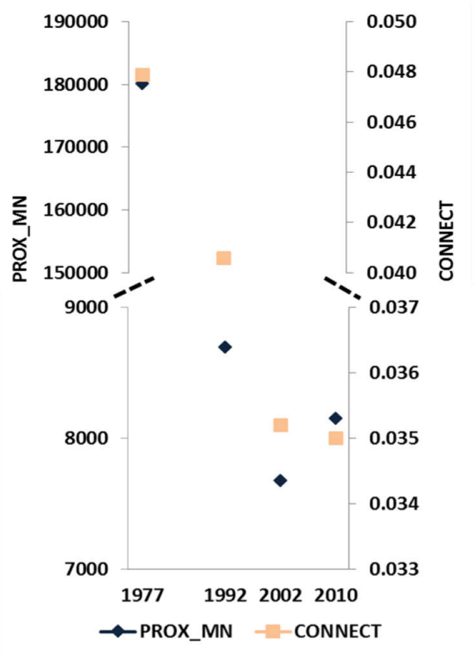

Figure IV-4: Connectivity indices at the patch (PROX_MN, MN = mean) and landscape (CONNECT) levels for the past study period. PROX index values are unit less and increase as patches become closer and are more contiguous (or less fragmented) in distribution and vice versa. CONNECT index: equals or is close to zero when the landscape is composed of a single patch or none of the forest patches are connected. This index equals 100 when all patches are connected. In our case, values are quite low because the landscape is composed of few, very big, well-connected patches and many small, poorly-connected patches.

4.2 Future forest change, fragmentation and connectivity

Our future scenarios showed substantial forest loss, which varied among scenarios depending on the assumptions about the amount of deforestation (Figure IV-5; Table IV-2). Should deforestation foreseen in the Forest Law only occur in green zones (scenario1_np), forest cover will shrink to only 65 % of the level at the end of the 1970s, or to 37 % if deforestation occurs in green and yellow zones (scenario2_np). If deforestation patterns follow trends from 1992 to 2010 (scenario3_np), forest cover will fall to 65 % of the extent in the late 1970s.

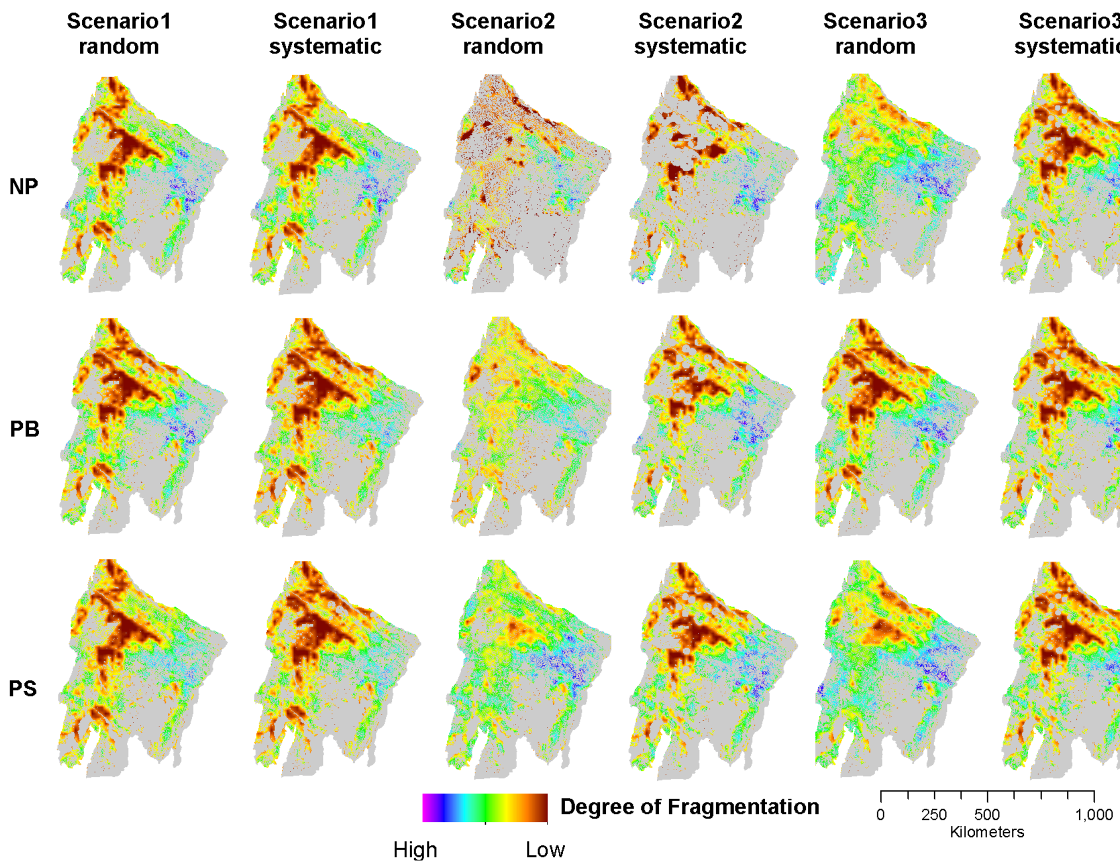

Figure IV-5: Scenarios of future forest extent and fragmentation degree for different levels of implementation of the Forest Law. NP = no protection, PB = protections of big patches only, PS = protection of big patches and stepping stones.

Our conservation strategies would lead to the preservation of 71 and 72 % of 1970s-level forest for both conservation strategies protecting big patches only, and big patches and stepping stones, respectively under scenario 1. For scenario 2, 59 and 64 % of the late 1970s-level forest would be protected by the strategies protecting big patches only, and big patches and stepping stones, respectively. For scenario 3, 67 and 68 % of the late 1970s-level forest would be protected by the strategies protecting big patches only, and big patches and stepping stones, respectively.

Analyzing future forest fragmentation showed that the implementation of the Forest Law in green zones under the protection of big patches and stepping stones (scenario1_ps*) would translate into the preservation of the highest amount of core forest (202,700 km^2 with 14,000 patches) across all scenarios (Figure IV-3). Fragmentation would be highest for scenario2_np, with a large increase in the number of patches (32,700), bridge forests (14,900 km^2) and island forests (9,400 km^2), and the lowest core area (104,000 km^2) among all scenarios. Conversely, extrapolating historic deforestation trends while protecting big patches and stepping stones (scenario3_ps*, Figure IV-3) would also lead to one of the lower forest fragmentation scenarios [core area: 174,000 km^2, 13,247 patches (Figure IV-3)]. Fragmentation components varied only marginally among the tenfold runs.

The degree of fragmentation was lower for scenario 1 but increased markedly for scenarios 2 and 3. Among these two scenarios, following a random allocation scheme resulted in higher degrees of fragmentation (Figure IV-5). The east of the Chaco experienced the highest degree of fragmentation for all scenarios due to the spatial distribution of forest in small fragments (i.e., on non-flooding areas) and to historically smaller agricultural fields in this area. The lowest degree of fragmentation was located in the northwest of the region, where big forest patches are situated.

Evaluating forest connectivity under alternative future scenarios showed that preserving big patches and stepping stones would maintain the highest connectivity when assuming lower amounts of deforestation. The overall highest landscape-level connectivity would be preserved for the scenario assuming deforestation in green areas under the protection of big patches and stepping stones (scenario1_ps*, Figure IV-6). Overall, preserving both big patches and stepping stones was the conservation strategy that maintained the highest degree of connectivity across all three conservation strategies, including when deforestation amounts were comparable among strategies such as in the case of (1) scenario2_ps*, scenario1_np* and scenario3_pb* or (2) scenario1_ps*, scenario3_ps* and scenario1_pb* (Figure IV-6). The strongest decrease in connectivity occurred under the currently planned implementation of the Forest Law (scenario2_np*), that is if green and yellow zones are not protected in addition to what is currently foreseen in the Forest Law (Figure II-6). However, the protection of big patches and stepping stones under this scenario (scenario2), would result in a substantial connectivity increase, despite relatively high amounts of deforestation (Figure IV-6).

4.3 Past and future forest loss in priority areas for conservation

Comparing past and future deforestation scenarios with the TNC conservation priority areas showed that in 2010, 45 and 55 % of the proposed final and terrestrial sets of priority areas, respectively, were still forested. The Forest Law fully protects only 15 % of these priority areas (i.e., in red zones), while 55 % can be sustainably used (i.e., yellow zones) and ~25 % are in green zones without use restrictions. The highest share of priority areas preserved among our scenarios occurred when big patches and stepping stones were protected and deforestation rates were not higher than in the past 20 years (scenario3_ps*, 95 % of the 2010 forest in the final set and 98 % in the terrestrial set preserved). In contrast, the lowest remaining forest area within priority areas occurred for the planned implementation of the Forest Law (scenario2_np, 52 % of the 2010 forest in the final set and 39 % in the terrestrial set preserved). Protecting big patches and stepping stones for this scenario would have a substantial effect, with 88 and 86 % of the 2010 priority-area forest preserved for the final and terrestrial set respectively. The scenario where deforestation would only occur in green areas while protecting big patches and stepping stones alike (scenario1_ps*) would also preserve 93 and 94 % of the 2010 priority-area forest for the final and terrestrial set respectively.

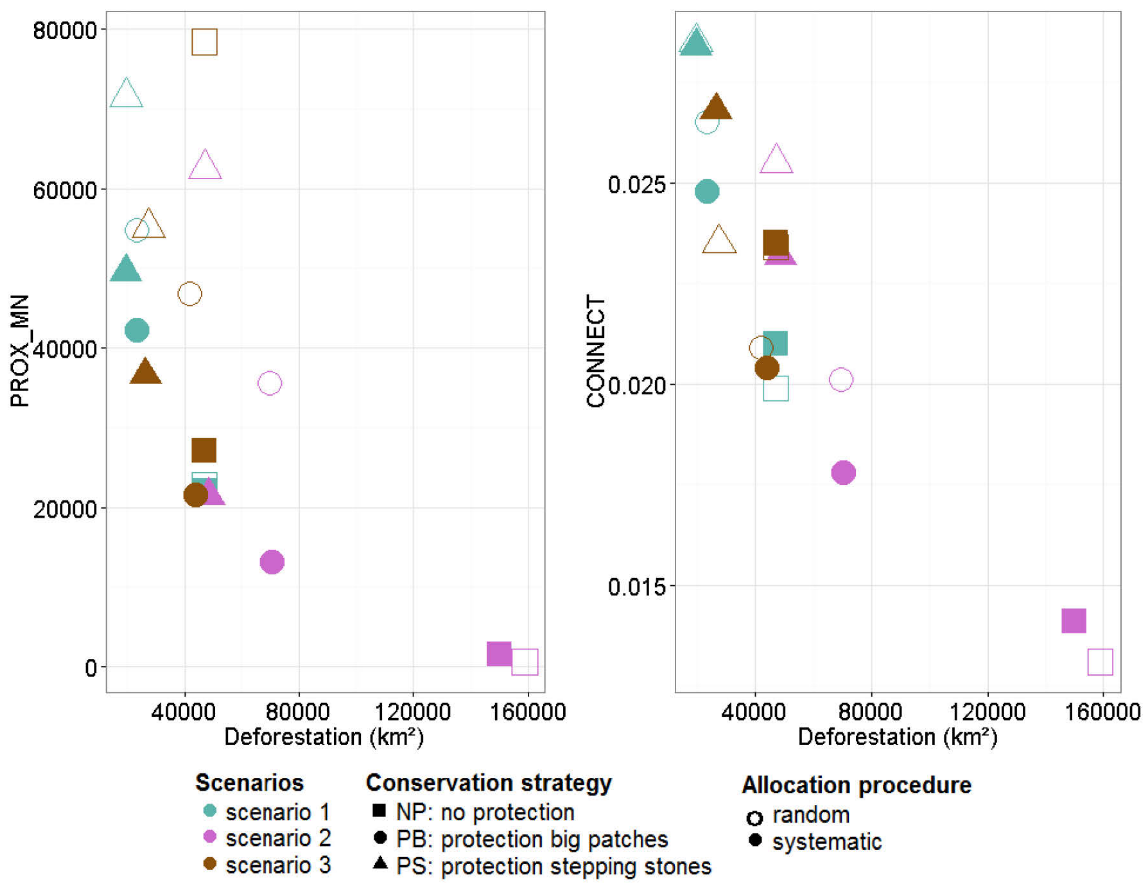

Figure IV-6: Connectivity at patch (PROX_MN) and landscape (CONNECT) levels versus deforestation for each scenario. Deforestation (in km^2, x axis) is in logarithmic scale.

5 Discussion

Compiling the provincial zoning plans for the Forest Law into an ecoregion layer clearly showed strong inconsistencies between provinces, including the same forest patch assigned to high-conservation value in one province (e.g., Santiago del Estero), but to potential deforestation in the neighboring province (e.g., Chaco). Moreover, our analyses showed that frameworks for how the Forest Law is implemented vary starkly between provinces. The provincial zoning maps were produced in different years and by differing institutions and stakeholders (i.e., universities, consultants, governmental departments, etc.). Provinces also interpreted and regulated the national Forest Law differently (Table SI IV-1). For example, up to 90 % of forests in green zones can be deforested in Chaco province, whereas Formosa allows deforestation only up to 60 % in the same zone. Similarly, red zones in Cordoba can be converted to some extent whereas red zones are strictly protected in all other provinces. These examples highlight the difficulties planners face in managing for forest connectivity at the ecoregion scale, and strongly suggest that provincial-level planning should be complemented by a broad-scale assessment of connectivity in upcoming revisions of the Forest Law.

We also found marked losses of high-conservation-value forest prior to the implementation of the Forest Law in areas that were later designated as red zones (where deforestation is prohibited), for example, in the provinces of Salta, Santiago del Estero and Santa Fe (AGN, 2014). Our results cannot attest to whether this deforestation occurred illegally. Deforestation might have been approved before the sanctioning of the law, yet there was a time window when future zoning was clear, but the Forest Law was not yet fully implemented (Gasparri and Grau, 2009; Seghezzo et al., 2011). The outcomes of the Argentine Forest Law to date, with increasing forest loss before the implementation of the law and continued fragmentation thereafter, are unfortunately not uncommon and resemble cases of increased forest loss prior to the implementation of nature protection in Brazil (Hardt et al., 2013) and Eastern Europe (Knorn et al., 2012; Kuemmerle et al., 2007). However, the Forest Law also undoubtedly had positive effects and lowered deforestation during the moratorium on land conversion issued by the Argentine government (2005–2006), when deforestation rates decreased in some provinces (Seghezzo et al., 2011).

The widespread and strongly-accelerating deforestation we documented resulted in a substantial increase in forest fragmentation and a loss of potential connectivity from 1977 to 2010 (Figure IV-3 and Figure IV-4). The particularly strong increase in fragmentation between 1977 and 1992 was likely due to widespread road building (e.g., Ruta 81) during that time (Ernst, 2014a). Further fragmentation was caused by a lack of coordinated planning and scattered conversion patterns that are a result of the small-scale ownership structure of the post-colonial smallholders' system (Adamoli et al., 2011). Together, this translated into the widespread creation of edge forest and the disappearance of bridges, especially in frontier regions, such as around Charata (Chaco Province). After 2002, connectivity loss was lower than in 2002, but this was likely due to a massive increase in forest patches due to more fragmented patterns in forest conversions than in the previous period, thus generating more stepping stones and hence slightly increasing connectivity at the patch level (Figure IV-3).

Our analysis of future scenarios reveals that, if all areas where deforestation is possible were converted, only 37 % of the 1977 extent of the Chaco's forest would remain. This percentage is close to thresholds frequently highlighted as critical for species' survival in fragmented landscapes (Camargo Martensen et al., 2012; Fahrig, 2003) and where edge effects could drastically change community structure (De Casenave et al., 1998). Given the increasing demand for food, feed, and biofuel, as well as the strong orientation of Argentine agriculture toward exports, further increases in deforestation are a plausible scenario. Still, our analyses suggest that additional, ecoregional conservation planning could effectively mitigate the outcomes for the region's forests connectivity and biodiversity.

Our fragmentation and connectivity results strongly emphasize the importance of forest patches functioning as stepping stones as key elements for maintaining landscape connectivity (Saura et al., 2013; Villard and Metzger, 2014). Stepping stones are particularly crucial in areas where climate change may lead to range shifts of species (Garcia et al., 2014; Gimona et al., 2015). Climate change has been marked in the Chaco in the past and is expected to continue to alter vegetation communities (Bravo et al., 2010; Ferrero et al., 2013; Murgida et al., 2014; Prado and Gibbs, 1993). Unfortunately, due to their small size and scattered distribution, stepping stones are often disregarded. This also seems to be the case in the Argentine Forest Law, where most stepping stones are assigned to green zones [e.g., remnants of "*bosque de tres quebrachos*" in the southwest of the Chaco province (Torrella et al., 2011), which would likely lead to their rapid loss without further protection.

Implementing conservation planning that would effectively maintain forest connectivity does not necessarily need to conflict with economic development. For example, our scenarios scenario2_ps*, scenario1_np* and scenario3_pb* resulted in approximate amounts of forest converted to agriculture, but the scenario that would include additional conservation planning to protect some big patches and stepping stones (scenario2_ps*), would lead to a substantial reduction in forest connectivity loss (Figure IV-6). Thus, low-cost/high-gain situations appear to exist in the Chaco, and smart conservation planning and landscape design (Moilanen et al., 2011; Turner Ii et al., 2013) that leverages these opportunities are needed to align agricultural production and conservation goals. We thus urge planners involved in the upcoming revision of the Forest Law to (1) mitigate the inconsistencies of the Forest Law design across provinces, (2) consider forest extent, fragmentation, and connectivity at the ecoregional scale, and (3) incorporate stepping stones as key landscape elements to preserve connectivity.

Ultimately, new protected areas are likely needed to safeguard the Chaco's forest in the future. Our analysis of conservation priority areas revealed that half of these areas will be lost if the Forest Law is fully implemented. However, the preservation of stepping stones would notably enhance the amount of priority areas safeguarded, even under the full implementation of the Forest Law. Currently, Argentina is the Latin American country with the lowest proportion of terrestrial protected areas [1.3 %, Figure SI IV-1,(Elbers, 2011)]. A useful approach to identify candidate sites for new protected areas would thus be to supplement the provincial-scale revisions of the Forest Law with an ecoregion-scale assessment of forest fragmentation and connectivity to identify those sites that would retain overall forest connectivity while preserving high-conservation-value areas.

While our sensitivity analyses highlight the robustness of our results, a few sources of uncertainty remain. First, we assumed that agriculture before 1997 only expanded into forested areas, which may be simplistic for some regions (e.g., Chaco province) where agriculture also expanded into grasslands. Second, we set a MMU of $0.1km^2$. To check that our choice of MMU did not bias our connectivity analyses, we calculated connectivity indices for a range of MMUs (i.e., 0.05, 0.1 and $0.2km^2$), showing that our results were robust across MMU scales. Nevertheless, we cannot fully rule out that smaller MMUs may lead to different connectivity results. Third, our connectivity analyses relied on simple, mainly structural indices; however, these have been shown to provide deep insights into regional-scale connectivity, similar to more complex functional connectivity analyses (Doerr et al., 2011; Ernst, 2014a; Ziółkowska et al., 2014). We also did not consider matrix quality, which can be important for preserving long-term landscape connectivity (Mastrangelo and Gavin, 2014; Ziółkowska et al., 2014). Although desirable, implementing functional and surface-based connectivity measures (such as circuit theory) across our scenarios would be challenging, due to the high number of forest patches and the overall large study area (500,000 km^2). Fourth, we used a search radius of 2 km for the connectivity and fragmentation analysis, representing maximum movement distances of intermediate dispersers. Sensitivity analyses showed that our results are robust towards the choice of this search radius (Table SI IV-2). Although a larger search radius may result in a more connected landscape (and vice versa), this would not affect the relative differences among our scenarios. Fifth, we used generic scenarios that entail simplified descriptions of the future. We consistently used the highest deforestation amount allowed in the provincial forest laws for the sake of comparability among provinces, but lower deforestation amounts may be enforced in some regions (e.g., in Formosa we homogenously used 60 % of deforestation in green zones when in some regions only 20 % is allowed, Table SI IV-1). Although considering socio-economic, demographic, technological, institutional or climate change effects in our scenarios would have been interesting, our goal here was not to identify plausible futures for the Chaco, but to explore the potential effects of the Forest Law under alternative implementations. Sixth, our allocation sampling strategies for potential deforestation (random and systematic) were applied in a mutually exclusive way, while a combination of both processes is likely the most realistic alternative. However, our results were robust across scenarios and against different allocation procedures (random and systematic), suggesting that our findings are independent from the actual conversion rate or mechanism assumed. Finally, our scenarios dealt with changes in forest only, while other ecosystems are also of high conservation value in the Chaco, such as natural grasslands (Macchi et al., 2013).

Managing for connectivity is challenging for large regions, where planning is often implemented at finer scales and the future effectiveness of conservation planning is uncertain. The implementation of the Forest Law in the Argentine Chaco highlights how sub-regional planning runs the risk of eroding connectivity at ecoregion scales, but also shows that considering overall connectivity can identify low-cost/high-gain options for land-use and conservation planning. Combining connectivity assessments with scenario analyses at scales most relevant for conservation and land-use planning is therefore important for drafting efficient conservation policies that are resilient against future environmental and socioeconomic change—in the Chaco and elsewhere.

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Supplementary Information

Table SI IV-1: Forest conversion allowed per province and zone in the Argentine Forest Law. Each provincial order specifies a range of forest conversion for each zone (e.g., 60-20% in green zones in the province of Formosa) sometimes according to some spatial attributes such as slope (e.g., Salta) or plot size (e.g., Chaco).

		Forest conversion (area %)		Nat. Law 26331/10 - Order 91/09	
Province	**Green zone**	**Yellow zone (productive and silvopastoral uses)**	**Red zone**	**Provincial order in council**	**Provincial law**
Chaco	90	70	0	932/2010	6409/2009
Santiago del Estero	70	60	0	1162/2008 and 1830/2008	6942/2009 and 6841/2007
Formosa	60	0	0	--	1552/2010
Salta	70	70	0	2785/2009 and 2211/2010	7543/2008 and 3136/11
Jujuy	100	25	0	2187-PMA-2008	5676/2010
Santa Fe	50	50	0	42/2009	in preparation
Cordoba	70	70	50	1476/2011/and 170/2011	9814/2010
Tucuman	95	50*	0	--	8304/2010
Catamarca	70	50*	0	1663/2011	5311/2010

Source: provincial orders and Adamoli et al. 2011. The percentages specified in this table under yellow zones consider productive and silvopastoral uses. (*) values set by the authors based on other provinces' percentages because forest conversion is possible but the provincial orders do not specify a percentage

Table SI IV-2: Evaluation of different search radii on the connectivity indices for scenario scenario1_np* (systematic allocation of conversions in green zones with no protection).

Radius [m]	**PROX_MN**	**CONNECT**
100	0.000	0.0000
1000	27231.911	0.0086
2000	31844.321	0.0229
5000	33392.425	0.0923
7000	33642.698	0.1620
10000	33794.529	0.2945
25000	33962.843	1.4393

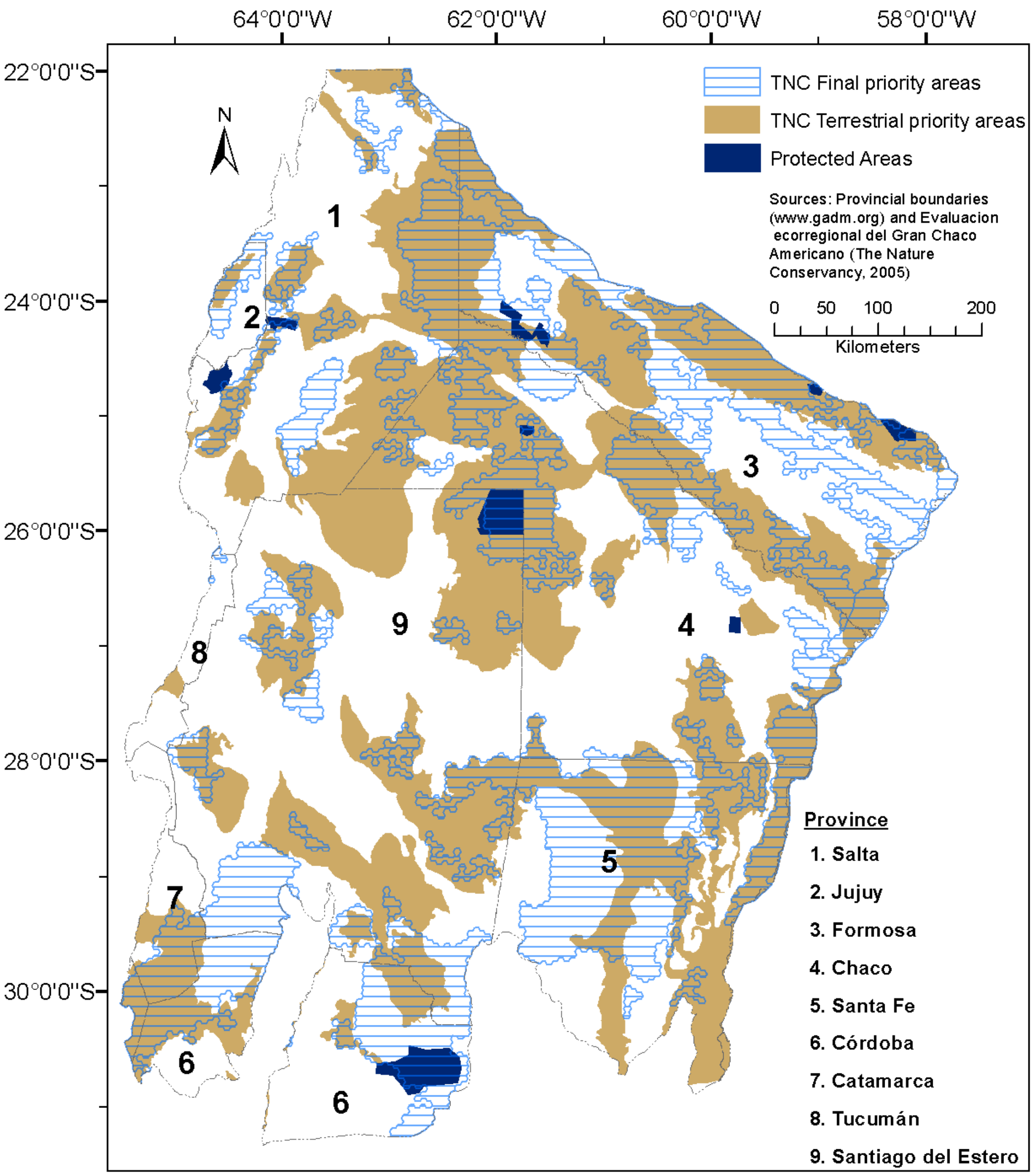

Figure SI IV-1: Study area with current protected areas and priority areas for biodiversity conservation of The Nature Conservancy (TNC).

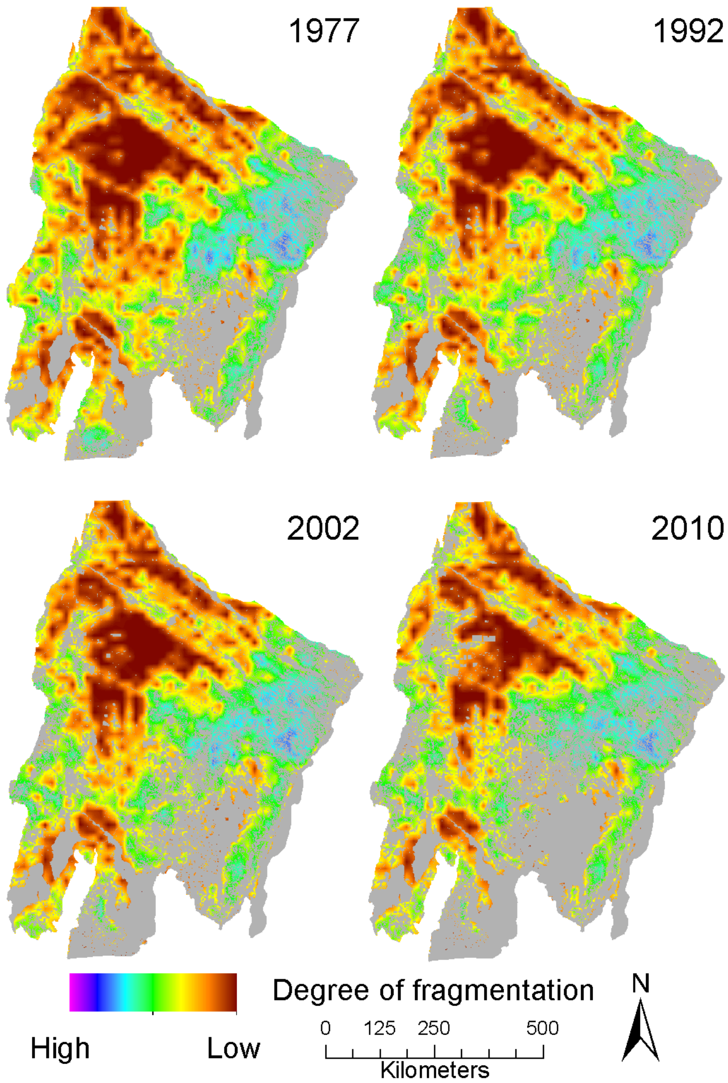

Figure SI IV-2: Degree of fragmentation (based on entropy theory) for the base years studied.

Chapter V:
Synthesis

1 Summary of main results from the thesis

The overarching goal of this thesis was to gain a better understanding of what drives land-use changes in northern Argentina and potential future land-change trajectories under different economic and conservation policies. Argentina is a country of drastic historic changes, arising from political, economic, and institutional settings, which strongly impacted Argentina's land system. Especially, developments in the agricultural sector have driven main land-use changes in Argentina, namely the conversion of forest, natural grasslands and savannas to cropland or grazing land and the intensification of grazing systems to more intensively managed cropland. These changes had severe impacts on Argentina's human-environmental systems. Studying the drivers of land-use change in the prime agricultural regions of Argentina (the Pampas, the Espinal, and the Chaco ecoregions) contributed to understanding the functioning of the regional land systems to assess the effects of economic and conservation policies by developing future scenarios of land-use change. The economic policies investigated in this thesis targeted the variation of agricultural profits and the improvements of the road network, both important prerequisites for agricultural developments. Besides, this thesis investigated how different levels of deforestation allowed by the Argentine Forest Law (26331) would maintain future landscape connectivity under different conservation strategies based on the preservation of different forest patch sizes. Furthermore, the environmental impacts resulting from the application of potential economic and conservation policies, as outlined in this thesis, were assesses and areas of conservation concern were highlighted. The insights gained from this research answered the three core research questions of this thesis.

<u>Research Question I:</u> *What drives agricultural expansion and intensification in northern Argentina?*

Chapter II estimated the probability of agricultural expansion (i.e., conversion of woodlands to either cropland or grazing land) and agricultural intensification (i.e., conversion of grazing land to cropland) for Argentina's prime agricultural regions as a whole. It also jointly quantified the importance of a range of underlying drivers and spatial determinants for these changes. Cropland expansion occurred mainly in areas of better agro-environmental conditions, whereas grazing land expansion occurred mainly in areas less suitable for cropping (e.g., soils of lower quality and drier climate) (chapter II). Yet, agricultural intensification was mainly sensitive to profit-related factors. This suggested that further agricultural intensification into the Argentine Chaco, is likely as long as agricultural demand and returns to agriculture continue to be high. However, results from this chapter suggested that non-market conditions, such as zoning or cultural ties to the land, could contribute to explain unexpected woodland conversions in the Chaco, where woodland in agricultural suitable regions was not converted probably because of indigenous presence or land zonation. This highlighted that economic policies (e.g., taxes or subsidies) are unlikely to alter woodland conversion rates and patterns dramatically and thus, zoning may be a more powerful tool steering land-system dynamics in the Chaco (chapter II).

<u>Research Question II:</u> *How may land use change under different economic and conservation policy scenarios in northern Argentina?*

Chapter III explored potential future pathways of land use in Argentina under different economic policies, namely changing agricultural profits and improving the road network. Thereby, it identified areas for which land-use changes are uncertain and where land-use and conservation planning efforts could be targeted. Chapter III showed that landscapes would experience an important change towards the intensification of agriculture and deforestation would slow down if land users search the maximization of profits from land. Assuming land-conversions would continue at 2000-2010 rates, agricultural expansion, predominantly for cattle ranching, would continue in the Chaco region, while agriculture intensification (i.e., grazing land to cropland conversion) would dominate in southern Chaco and the Pampas. Whereas economic policies targeting higher profits influenced the concentration of agriculture intensification, policies targeting lower profits affected expansion rates of grazing lands markedly. The later resulted in moderate concentration of cropland and a lessening of deforestation pressure in marginal regions. Improving the region's road networks resulted in further expansion of cropland and grazing land into remaining woodlands and natural grasslands. Overall, chapter III suggested that economic policies targeting profits (e.g., taxes, subsidies, Payment for Ecosystem Services) have restricted influence in land use changes other than often assumed. Given that results from chapter III also showed continued and high conversion pressure on the region's remaining natural vegetation, zoning is likely a more powerful tool to avoid unwanted outcomes of the ongoing agricultural expansion and intensification trends in the Chaco (as already inferred from chapter II).

Chapter IV analysed the potential effect of the Forest Law on forest loss due to agriculture expansion (i.e., cropland and grazing) and forest fragmentation and connectivity in the Argentine Chaco under forest conservation strategies. This chapter highlighted that the full implementation of the Argentine Forest Law as planned, without any other conservation strategy, would drastically decline forest area and landscape connectivity in the Chaco. Results from this chapter also showed that agricultural development can be balanced with the maintenance of forest connectivity when conservation policies and spatial planning are in place. The conservation strategy that protected forest patches, acting as stepping stones, showed the potential to substantially mitigate connectivity loss due to deforestation, meanwhile allowing agricultural developments according the Forest Law zonation. This chapter showed how land zonation coupled with conservation strategies can aid informing conservation policy to balance agricultural activities and natural systems. This is important for the upcoming revisions of the Argentine Forest Law, and, more generally, in debates about sustainable resource use.

Research Question III: What are potential environmental impacts of future land-use changes in northern Argentina?

Chapter III and IV evaluated the potential impacts of economic and conservation policies and zoning (Forest Law) on natural ecosystems in northern Argentina (Pampas, Espinal and Chaco). This was done by first quantifying the potential loss of natural vegetation under future land-use (chapter III and IV). Second, fragmentation and connectivity changes under potential future land-use was evaluated (chapter IV) and finally, impacts from land-use scenarios on priority areas for conservation were assessed (chapter III and IV). Under the full implementation of the Forest Law and extrapolating deforestation rates in 1990-2010, 65% of the forest cover in the late 1970's could be lost and connectivity would drastically decrease (chapter IV). Likewise, if deforestation rates in 2000-2010 are extrapolated into the future, 33,5% of the forest in 2010 could be converted in Argentina until 2030 (chapter III). This increase in deforestation rates were due to deforestation doubling since the 1990's (chapter IV). Yet, if landowners seek maximizing land profits, deforestation may slow down in the future (chapter III). Moreover, if spatial planning in Argentina does not consider the loss of forest connectivity from further agricultural developments in the design of conservation policies, half of the Priority areas for conservation identified by The Nature Conservancy (TNC) could be lost in the future and future forest connectivity could decline (chapter IV). Likewise, if spatial zoning fails to limit agricultural expansion in designated areas, valuable natural grassland areas with high importance for conservation could be substantially affected (chapter III). This could potentially translate into important socio-environmental trade-offs that could affect local livelihoods and ecosystem services in the region. Therefore, integrating an ex-ante evaluation of environmental impacts from potential further agricultural development in Argentina can benefit the spatial planning of human-environmental systems to avoid unwanted outcomes from economic and conservation policies.

2 Cross-cutting insights from the thesis as a whole

Based on the results from the individual research questions, five cross-cutting insights emerged from this thesis.

First, understanding the drivers of specific land-use conversions is crucial to better inform decision makers (chapters II and III). Results from this thesis highlighted that agricultural intensification was more responsive to profits meanwhile agricultural expansion was more dependent on environmental conditions and zoning (chapter II). Differentiating drivers of agricultural expansion from those of intensification was possible because this thesis analysed land-use conversions (such as woodland to grazing land or cropland) instead of only land use/cover. Yet, the definition of intensive agriculture can be region and user specific. Current efforts are being made in homogenizing a coherent definition of land-use intensity within the land-use science community. The new working framework focuses on land systems instead of on specific land-use/cover conversions (Levers et al., 2015; Václavík et al., 2013; van Asselen and Verburg, 2012). The strength of this new approach lies on the integration of the multidimensional aspects of land-cover changes with land-use intensity and land management, by means of including agricultural input and output usage for example, or changing degrees of mechanization (Erb et al., 2013). The spatial design and patterns of land systems (i.e., architecture) is also an influential factor in understanding potential land systems trajectories for land use sciences (Turner Ii et al., 2013). This new framework improves the understanding of human-environment interactions providing a more accurate representation of the relationships of socio-ecological systems (van Asselen and Verburg, 2012). Moreover, chapter II suggested that zoning policies may be more efficient in curbing deforestation meanwhile economic policies may be more efficient in steering agricultural intensification and this was confirmed in chapter III. Chapter III showed that policies affecting agricultural profits translated into the further intensification of agriculture and less cropland expansion in marginal regions. The efficiency of zoning in curbing deforestation in commodity frontiers has been recently proved for Argentina in an collaborative study developed during the course of this thesis (Nolte et al., 2017a) and for South America (Nolte et al., 2017b).

Second, land users responsiveness to economic policies are path dependent (Chavez and Perz, 2013) and sometimes even drastic economic policies can have less effects than expected in land conversions (chapter III). Limited effects of economic policies can be explained, on the one hand, by market price feedbacks, since a policy intervention that rises the profit of one land use can rise indirectly the profits of other land-uses and thus the total effect of the incentive itself is "diluted" in the total market gain (Lawler et al., 2014; Radeloff et al., 2012). On the other hand, other factors not related to profit maximization, such as cultural ties to the land, land speculation, capital accumulation or path dependencies (Chapter II) impose a slow reaction or a delayed response (i.e. inertia) to a policy incentive due to land-users' limiting factors (such as capital availability) or preferences (such as subsidies beneficiaries or social values) (Chavez and Perz, 2013; Garrett et al., 2013; Henderson et al., 2013). Thus results from this thesis contribute to challenge some of the fundamental theories of classic land rent economy that assume the preference of landowners to maximize land utility and thus investments in land developments that maximize landowners' rent (Bockstael et al., 2000). It also contributes to the notion of inertia (sensu Reenberg, et al. (2012)) since as shown in Chapter III, some of the most northern regions in Argentina that experienced infrastrusture developments at the end of the study period when the NRM was parametrized (2010), were not depicted as dynamic areas in the scenarios of chapter III, which points into a delay in the reponse to economic incentives in more marginal regions that are not agricultural clusters (Porter, 1998).

Third, policies can benefit from decentralization and being context specific in large regions. Centralized policies can have divergent and leakage effects when applied to heterogeneous large regions and for example, the Forest Law in Argentina is a good example of how decentralization can function in curbing deforestation (Nolte et al., 2017a). Chapter III showed how economic policies targeting decreasing agricultural profits for the Chaco and the Pampas regions, can maintain forest cover in marginal areas but may translate into grazing land expansion in other areas more responsive to profits. Therefore, chapter III highlighted that there is not such a one-policy-fits-all approaches and thus the need for decentralized and spatially context specific decision-making, especially in agriculture frontiers where the heterogeneity of the land dynamics and its users require more tailored approaches. This claim is also being done for other large and heterogeneous regions in the world where policies are centralized but land system dynamics are region specific (Levers et al., 2015).

Fourth, the coupling of several policy incentives can lead to unexpected effects (Hennessy, 1998; Zhang et al., 2014). For example, chapter III showed how economic incentives coupled to policies increasing agricultural yields, could translate into the deforestation of sensitive regions that would be preserved otherwise when only economic incentives are in place. Yet, positive effects from coupled policies are also possible and, as chapter IV showed, under similar deforestation rates according to the zoning of the Forest Law, additional conservation strategies had the power to steer and maintain landscape connectivity without compromising agricultural development. This suggests that the implementation of policies should be done with care and with a priory evaluation of potential impacts of coupled policies and their accumulated effects (Helming et al., 2008).

Fifth, potential environmental trade-offs or conservation opportunities can be detected by ex-ante assessments of scenario analysis' results that evaluate potential policy impacts (Helming, 2014). Both chapters III and IV showed how potential future land-use conversions could affect priority areas for conservation (Bilenca and Miñarro, 2002; TNC, 2005) if future agricultural developments are not planned with the goal of balancing land-use changes and environmental conservation. Particularly, chapter IV detected 45% of the cover of priority areas for conservation of the TNC, as hotspots of potential conversions, where future deforestation could seriously affect the natural vegetation of priority areas and where urgent protection is required (Myers et al., 2000). Likewise, chapter III detected 19 priority areas for conservation in forest and natural grasslands of Argentina that could be highlighted as potential hotspots of future land-use conversions if past land-use trends are extrapolated into the future. The concept of potential future hotspot of land-use conversions builds on the notion of Kuemmerle et al. (2016) that detect regions where land-use change concentrates, although the latter is defined based on land-use intensity metrics. Contrary, other areas priority for conservation would not experience conversions under any scenario (chapter III), such as the east of Salta, north of Chaco or west of Formosa. Coldspots of conversions that are located within priority areas for conservation can potentially be used to focus conservation investments away from these regions, since they are less likely to experience conversions in the future. Yet, these areas may be home to important ecosystem services and to specialist species since they do not experience high land-use dynamics and thoughtful planning is needed (Kareiva and Marvier, 2003). There were some priority areas for conservation that would be affected only under some scenarios, such as in the south east of Salta or Santa Fe (chapter III), where careful planning is necessary since these areas may be particularly sensitive to small changes in policy incentives. Additionally, since this thesis highlights the importance of maintaining the connectivity of landscapes (chapter IV), efforts that prioritize the protection of new areas in the face of global change (Venter et al., 2014), should be invested in a connected network of potential future hotspots of land conversions located in priority areas for conservation (i.e., areas of conservation concern in chapter III) (Marchese, 2015).

Moreover, chapter IV showed that informed spatial planning can influence landscape connectivity, because under similar deforestation pressure, conservation management options improved landscape connectivity in the study area. This chapter also contributed to challenge traditional paradigms in (landscape) ecology that highlight the importance of large patches of vegetation (versus small patches) supporting a higher species persistence by allowing species to disperse and adpat to environmental changing conditions among others (Ferraz et al., 2007; MacArthur and Wilson, 1967). Yet, small patches of vegetation that remain in the agricultural matrix play an important role to maintain ecosystem functionality and species persistence at the landscape scale since they allow for dispersal and better adaptation to global change (Saura et al., 2013; Tulloch et al., 2016). Results from this thesis also supported the importance of small patches in maintaining landscape connectivity (chapter IV) thus suggesting their contribution to species dispersal and persistence.

3 Conclusions and implications

Although dynamic agricultural regions in Argentina often respond to drivers of land-use change as expected (i.e., spatial agricultural suitability and economic underlying forces trigger agricultural developments), agricultural intensification and expansion may be influenced by other factors that need specific consideration in spatial planning actions. Sometimes, agriculturally dynamic regions do not respond to economic incentives as expected and social aspects or agglomeration economies may play an important role in steering development pathways and thus preserving ecosystems and their associated services in the Chaco and the Pampas ecoregions. Since future agricultural development in northern Argentina threatens areas valuable for conservation, the current zonation in Argentina (Forest Law) has the potential to lower the environmental trade-offs from agricultural developments. In this sense, the often forgotten small patches of natural areas that remain in the agricultural matrix can play a key role in maintaining landscape connectivity. Yet, implementing conservation planning that would effectively maintain forest connectivity does not necessarily need to conflict with economic development as shown in this study. Moreover, small patches of natural areas can serve to preserve a more heterogeneous landscape matrix that enhances the availability of natural resources for local livelihoods and thus facilitate species permanence in dynamic agricultural regions.

Human-environmental systems are intricately related and so are the impacts that human activities have on these systems. Therefore, environmental trade-offs arising from potential future land-use conversions have a direct effect on local livelihoods and small family farms that benefit directly from the availability of local resources in their immediate surroundings. This does not mean that higher levels of protection may lower environmental impacts and thus secure resource availability for local inhabitants, as it has been refuted (Sunderland et al., 2007), but that cautious planning for balancing socio-environmental trade-offs when designing spatial policies is of critical importance to avoid unwanted outcomes from policies in dynamic agricultural regions.

4 Outlook

Overall, this thesis showed the potential of understanding the effects of drivers of land-use change under different policies to better inform policy design for steering land systems into desired pathways. Argentina was a perfect example of a complex human-environmental system that was used to evaluate the utility of scenarios of future land-use conversions under different economic and conservation policies in agricultural dynamic regions. During the course of this thesis, Argentina changed its government, and is now giving high priority to expanding the transportation network and to improving the local livelihoods of rural populations in the north of Argentina (*Plan Belgrano*). This may increase the accessibility of these region compared to the assumptions realized in our scenarios. Consequentially, an even stronger increase in agricultural expansion due to lower transportation cost and easier access to previous marginal lands can be expected. At the same time, better social conditions due to economic benefits may arise from these developments, which can improve life-standards of local livelihoods and support the development of rural areas. This potential increase of rural population was also not foreseen in the scenarios of this thesis and may translate into higher agriculture expansion and stronger environmental impacts than those assumed. Further scenario analysis should thus incorporate projections of population development, migrations into or out of the region, and associated potential expansion of agricultural activities (i.e., rebound effect). In addition, there were important developments in Argentina towards the production of crop for biofuels generation (INTA PNB program- *Programa Nacional de Bioenergía*). This was partially boosted due to the old EU regulation that decided promoting biofuels for energy generation (Renewable Energy Directive 2009/28/EC). The new amendment to this EU Law (Directive EU 2015/1513), that focuses on *advanced biofuels* (non-food crops and other vegetable resources with low GHG emissions), may lead to the disappearance of a large market for Argentina to export biofuels and may lower the expansion of biofuel crops in Argentina.

The integration of economic and ecological modelling is also an important future research direction to overcome information challenges in policy design (Brady and Irwin, 2011; Duke and Wu, 2014). The results from scenarios of future land-use conversions, as developed in this thesis, can be integrated into sophisticated spatial conservation priorization tools to better target regions that can experience future developments without further compromising ecosystems of high value for conservation. Conservation planning recognizes the existence of competing land-uses that need to be balanced during the planning stage to avoid environmental trade-offs (Moilanen et al., 2011). Spatial priorization tools can be used to balance competing land-uses according to development goals. Often this can be done by zoning the land under the land-use category that minimizes environmental trade-offs and opportunity costs for conservation therefore enhancing sustainable multifunctional landscapes (Lewis and Nelson, 2014; O'Farrell and Anderson, 2010). However, current spatial priorization tools generate scenarios based on policy options that are within the system boundaries of study. However, the integration of external results from land-use change scenarios would result in a more complex but also more holistic priorization approach, opening a door to a new integrated conceptualization of human-environmental systems. For example, results from this thesis could potentially highlight the role of zoning in such priorization exercise, generate future land-uses based on economic and conservation policy options and delineate areas of conservation concern that can be masked from future developments (chapters II, III and IV). These external results can serve as new input for spatial priorization tools or in participatory planning exercises with stakeholders. Results from this thesis thus provide important information for spatial and conservation planning since they can aid in informing policy design and decision making (Gavier-Pizarro et al., 2014).

This thesis leaves some room for improvements of the research methods applied in this work, on which potential further research can be based on. First, the availability of agricultural production and land-use/cover data with higher spatial and temporal resolution would have allowed to better characterize important political changes in Argentina, such as the implementation of the Forest Law, and their impacts on land conversions. Also, the analysis of functional landscape connectivity that requires species specific friction movement maps based on landscape matrix characteristics could be improved when such land-use maps become available. However, in the course of this thesis a more detailed land-use change dataset was developed for the Gran Chaco that would allow for more sophisticated connectivity analysis (Baumann et al., 2016). Evaluating the influence of conservation management actions in landscapes of different spatial configurations (e.g., maintaining hedgerows between agricultural plots vs. homogenous agricultural landscapes) would be an important future research topic, too. Different spatial planning and conservation actions that translate into different structural land-use/cover patterns can have an effect on landscape heterogeneity, connectivity and potentially, on species persistence (Balmford et al., 2012; Stein et al., 2014). Moreover, expanding the study area to the entire Chaco (Argentina, Bolivia and Paraguay) would enrich the understanding of the agricultural intensification and expansion processes and associated deforestation in one of the biggest dry forest ecoregions of the world in need of conservation (Kuemmerle et al., 2017). Gathering the spatial and statistical data for modelling and scenario generation for the three bordering countries should be subject to future research. The methods applied in this thesis offer a robust framework for expanding this type of study to other dynamic agricultural regions of the world.

Land-use changes in agriculturally dynamic regions face many uncertainties regarding the extent and location of future changes (Laurance et al., 2014; Meyfroidt et al., 2014). Increasing globalization and future human demands will influence land-use trajectories but they can be steered by policies and managed by widely informed spatial planners. A deeper understanding of the drivers of land-use change and how policies can impact human-environmental systems bring opportunities for spatial and conservation planning to steer future development pathways towards desired directions, in the global South and elsewhere.

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Publikationen

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[8] Nolte C., Gobbi B., Le Polain de Waroux Y., **Piquer-Rodríguez M**., Butsic V., Lambin E.F. (2017) Decentralized land use zoning reduces large-scale deforestation in a major agricultural frontier. Ecological Economics 136, 30-40.

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PEER-REVIEWED BOOK CHAPTERS

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Eidesstattliche Erklärung

Hiermit erkläre ich, die vorliegende Dissertation selbstständig und ohne Verwendung unerlaubter Hilfe angefertigt zu haben. Die aus fremden Quellen direkt oder indirekt übernommenen Inhalte sind als solche kenntlich gemacht. Die Dissertation wird erstmalig und nur an der Humboldt-Universität zu Berlin eingereicht. Weiterhin erkläre ich, nicht bereits einen Doktortitel im Fach Geographie zu besitzen. Die dem Verfahren zu Grunde liegende Promotionsordnung ist mir bekannt.

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Berlin, den 20.02.2017